RAPID INFILTRATION OF PRIMARY SEWAGE EFFLUENT
AT FORT DEVENS, MASSACHUSETTS

COLD REGIONS RESEARCH AND ENGINEERING
LABORATORY, HANOVER, NEW HAMPSHIRE

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Rapid infiltration has provided final treatment to unchlorinated Imhoff tank effluent at Fort Devens, Massachusetts, since 1942. Wastewater flow has varied seasonally; however, most flows to the 22 treatment beds at the installation in 1973 were 2676 to 9541 m^3/day (1.3 million gallons per day). In an operation cycle of simultaneous inundation of three 0.31-hectare treatment beds for 2 days, followed by a 14-day recovery period, effluent application has been about 27.1 m/yr. Chemical analyses of soil samples obtained from the upper 3.05 m of the treatment beds showed that levels of organic matter, electrical conductivity, nitrogen, phosphates, calcium, magnesium, potassium, manganese, sodium, copper, iron, and zinc were substantially higher than those of background samples, whereas

20. Abstract (cont'd)

levels of sulfur, chloride and cation exchange capacities were only slightly higher. Treatment bed soil samples had lower pH levels and boron concentrations than those found in background samples. The quality of the primary effluent applied to the treatment beds and the groundwater in 14 observation wells was determined by sampling at biweekly intervals and analyzing the samples for pH, conductivity, alkalinity, hardness, five-day biochemical oxygen demand, chemical oxygen demand, total nitrogen, organic nitrogen, ammonium nitrogen, nitrate nitrogen, nitrite nitrogen, total phosphorus, orthophosphorus, chloride, sulfate and total coliform bacteria. Groundwater quality in wells located 60 to 100 m from the application area showed that the primary effluent after flowing through the sand and gravel formation, had been substantially renovated. Before treatment, effluent COD and BOD_5 levels in the groundwater were about 10 to 20 mg/l and 1.0 to 2.5 mg/l, respectively. Concentration of total nitrogen in the primary effluent was 47 mg/l, which was divided equally among organic and ammonium forms. Total nitrogen, primarily as nitrate nitrogen, in water samples from observation wells affected most was 10 to 20 mg/l. This was a reduction of approximately 60 to 80% in total nitrogen levels. Changes in the concentration form of nitrogen showed that applied nitrogen was nitrified, with substantial nitrogen removal due mostly to denitrification. Phosphorus levels in the primary effluent averaged 11 mg/l, 80% of which was orthophosphate. Total phosphorus levels in the groundwater were 1.0 to 2.0 mg/l, slightly above the 0.7 mg/l background level. Orthophosphate was generally less than 0.4 mg/l in wells located around the treatment area but was higher than the 0.1 mg/l background level. Chloride, sulfate, pH, and hardness levels varied little from observed effluent levels, although conductivity and alkalinity were substantially less. Analysis of groundwater samples of fecal coliform bacteria proved negative. Mean numbers of total coliform bacteria in the groundwater were less than 200/100 ml, although two wells showed mean counts of 400 to 600 organisms/100 ml of samples.

PREFACE

This report was prepared by Melvin B. Satterwhite, Agronomist, New England Division, Corps of Engineers; Gordon L. Stewart, Soil Scientist, Department of Plant and Soil Science, University of Massachusetts, Amherst, Mass.; Brian J. Condike, Chief Chemist, Water Quality Laboratory, New England Division, Corps of Engineers; and Edward Vlach, Chief Chemist, Massachusetts Agricultural Experiment Station, University of Massachusetts, Amherst, Mass.

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SUMMARY

Investigations conducted at the Fort Devens, Massachusetts, wastewater treatment facility during 1973 showed that rapid infiltration can effectively renovate unchlorinated primary effluent to quality better than that achieved by conventional tertiary wastewater treatment in a northern environment. Treatment of the 5,049 m^3/day domestic sewage was accomplished using Imhoff tanks and 22 rapid infiltration treatment beds. About 27.1 m of effluent was loaded on the treatment beds. Analysis of soil samples from surface horizons in the 60-m thick deposit of stratified sands and gravels showed substantial increases (200 to 500%) in the levels of nitrogen, phosphates, calcium, magnesium, potassium, sodium, and iron over background concentrations. Levels of copper (about 10 to 50 ppm) and zinc (about 20 to 60 ppm) were also greater than background levels.

Biweekly sampling and analysis of primary sewage effluent and groundwater in 14 observation wells showed that primary effluent had been renovated to an extent comparable to that achieved by conventional tertiary systems. Groundwater COD and BOD_5 were about 10 to 20 mg/l and 1.0 to 2.5 mg/l, respectively. Organic nitrogen and NH_4-N were generally less than 4 mg/l in the groundwater adjacent to the disposal site; effluent concentrations were 11 to 33 mg/l and 6.2 to 42 mg/l, respectively. Effluent NO_3-N varied from 1 to 2 mg/l during 1973, but was 10 to 20 mg/l in the groundwater of several wells. Comparing mean total effulent nitrogen with groundwater concentration over the study year revealed that groundwater contained only 21 to 43% of the nitrogen observed in the effluent. However, the greater portion of the 10 to 20 mg/l total nitrogen remaining was NO_3-N, which was in concentrations generally exceeding those allowed by drinking water standards for municipal waters.[19]

Total phosphate in the groundwater was about 2 mg/l, 0.2 mg/l of which was orthophosphate. Total phosphate in the groundwater was reduced 80% in relation to the levels in the primary effluent. Chloride and sulfate levels were not substantially affected by the filter medium.

Infrequent determinations of fecal coliform bacteria in groundwater samples proved negative, although substantial numbers were present in the primary effluent. Mean total coliform bacteria levels in the groundwater were about 200/100 ml, but the mean number in the effluent was 32×10^6/100 ml.

Results of these investigations show that the treatment beds have renovated the primary sewage effluent to levels much better than those achieved with conventional secondary treatment. Impact of the percolate on the groundwater quality in the background area adjacent to the disposal site has increased the levels of most constituents. However, the groundwater adjacent to the site, with the exception of chloride, nitrate, and ammonia levels, still meets drinking water quality standards.

The results of this study will enable a more detailed assessment to be made of the feasibility of using land treatment as an economical alternative to other wastewater treatment methods, while still achieving water quality comparable to that obtained by conventional tertiary wastewater treatment facilities. We believe that, as a result of this study, military facilities commands that must treat wastewater may better evaluate the land treatment alternative, based on experience at an existing military installation.

RAPID INFILTRATION OF PRIMARY
SEWAGE EFFLUENT AT
FORT DEVENS, MASSACHUSETTS

M.B. Satterwhite, G.L. Stewart,
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INTRODUCTION

Use of the biological, chemical and physical process-
es of the soil in the treatment of municipal wastewater
is an alternative to use of conventional secondary and
tertiary wastewater treatment facilities. Spray irriga-
tion, overland flow, and rapid infiltration have been
used to treat wastewater in the past. Each approach
has different design and operation criteria which de-
pend on site characteristics and other local factors.
Where naturally occurring sand and gravel deposits
provide a workable depth, the use of rapid infiltration
on these materials may be an efficient method for treat-
ing municipal and industrial wastewater.

With proper operation and maintenance, rapid in-
filtration beds have provided long-term service with
substantial wastewater renovation.[7 8 10 11 15 23] Im-
portant to maintaining renovation capability is the
operational cycle during which the treatment bed is
alternately inundated and then permitted to rest. The
inundation period varies depending on geographical
location, soil characteristics and climatic conditions.
Regularly scheduled intermittent rest periods for the
treatment beds are necessary to permit recuperation of
infiltration rates which degenerate over the inundation
period because of physical and biological clogging of
the soil.[3 8]

Physical clogging of the soil results from the trans-
port and deposition of particulate matter in the treat-
ment beds. Lodging of suspended solids, usually of an
organic nature, between soil particles leads to partial
or complete blockage of soil pores, thereby decreasing
substantially the effective pore size, and thus reducing
the infiltration rate. Eventually, an organic layer
forms on or near the surface horizon. Once formed,
this layer reduces oxygen transfer into the soil and
decreases the effectiveness of the soil system in reno-
vating wastewater.

Biological clogging of the soil also results from
microbial growth on the filtered organic matter and
on soil particles. McGauhey and Krone[18] concluded
that various microorganisms and microbial byproducts
were responsible for the biological clogging of the soil
system. Fortunately, during the rest period, biological
and chemical processes reestablish most of the soil's
original infiltration capacity. Biological utilization of
filtered organics is dependent upon the nature of the
organic materials and oxygen availability. Where
ample oxygen is present, the easily biodegradable sub-
stances, such as sugars or starches, are readily assimi-
lated; but tannins, lignins and waxes, which are not
easily degraded, tend to concentrate.[15 18]

Because reclaimed waters from land treatment sys-
tems often recharge groundwater aquifers or surface
water bodies, the quality of renovated water is an
important consideration. Laverty et al.[15] found that
rapid infiltration and natural soil adsorption of pre-
treated wastewaters resulted in a high-quality water
suitable for groundwater recharge. Bouwer[7] observed
that rapid infiltration of wastewaters that had received
secondary treatment removed essentially all suspended
solids, biochemical oxygen demand (BOD) and fecal
coliform bacteria. The concentration of phosphorus,
about 13 ppm in the secondary effluent, was reduced
to 5 to 7 ppm after infiltration. Similarly, Amramy[3]
observed a significant reduction in BOD and phosphate

levels, while chemical oxygen demand (COD) was reduced from 172 ppm to 85 ppm.

Nitrogen levels in applied effluents, and subsequently in waters reclaimed from land treatment systems, remain a major consideration where these waters augment public water supplies. Nitrogen from primary sewage effluents is mostly organic and ammonium nitrogen, while nitrogen in secondary effluents is predominantly in the ammonium form. Laboratory and field investigations under aerobic conditions have shown that organic nitrogen and ammonium nitrogen applied to treatment beds are readily converted to nitrate.[8][12][13] However, not all the nitrogen applied finds its way into underlying groundwater. Bouwer[7] observed a 70% reduction of nitrogen in reclaimed water in comparison with nitrogen in the applied secondary effluent.

Mechanisms for possible nitrogen removal have been discussed by Bouwer[7] and Lance.[12] Volatilization of ammonium from wastewater effluents could remove some nitrogen; however, because ammonium volatilization is pH dependent, significant amounts of ammonium would not be volatilized unless the effluent pH were made more alkaline by lime additions. Adsorption of ammonium by the negatively charged clay and colloid particles of soil could also remove considerable amounts of ammonium, but the quantities of ammonium added to the rapid infiltration beds in the effluent would saturate the adsorption capacity in a few years.

Biological uptake and incorporation of nitrogen into cell tissue could remove substantial nitrogen, but unless the biological material were removed from the application area, the nitrogen would be released back to the system upon microbial decay of the cell tissue. Ammonium fixation by organic complexes could remove large quantities of nitrogen, but the fixation mechanism has a limited capacity which would be satisfied after a short time by the large nitrogen additions to the rapid infiltration system.

Biodenitrification in rapid infiltration systems is possibly the most important nitrogen removal mechanism. Bouwer[7] observed that nitrogen removal by denitrification is distinctly possible in land treatment systems where oxygen is limited and both nitrate and organic matter are present. By this mechanism, nitrate is converted mainly to nitrogen gas which escapes into the atmosphere. Although the denitrification process has not yet been positively demonstrated in rapid infiltration systems, substantial nitrogen removal has been attributed to it.[11][17]

Field studies of rapid infiltration systems have been conducted on new facilities. These investigations have demonstrated that the wastewater treatment technique can provide acceptable renovation of wastewater on a short-term basis. However, questions pertaining to the long-term effectiveness of rapid infiltration systems for treating primary sewage effluents still remain.

The objectives of the present study were to determine the effectiveness of a rapid infiltration system in renovating unchlorinated primary sewage effluent after more than 30 years operation in a northern climate, and to ascertain any physical and chemical changes in the soil of the treatment beds. Investigations were begun in October 1972 and were completed in January 1974.

DESCRIPTION AND OPERATION OF TREATMENT FACILITY

Fort Devens is a U.S. Army military installation located about 52 km northwest of Boston and 35 km northeast of Worcester, Massachusetts. The post includes portions of four Massachusetts communities: Ayer, Harvard, Lancaster and Shirley. The number of persons residing or working on the post has fluctuated with the mission of the installation. In 1973, the daytime population was about 15,000, 10,400 of which were permanent residents.

The present sewage treatment facility was constructed in 1942 (Fig. 1). From 1 September 1946 to 31 August 1948, when military activities were curtailed and the post was used as the Fort Devens Extension of Massachusetts State College at Amherst (now the University of Massachusetts), the sewage treatment facility continued to provide service. Sewage from the cantonment and housing area is carried in sewerlines by gravity flow except for several small pumping stations required to lift the wastewater at several points on the post. By the time the sewage is transmitted to the main pumping station, kitchen grease and fats and various oils have been removed in grease traps. These traps are cleaned periodically and the materials collected are deposited in sanitary landfills.

At the main pumping station, the sewage passes through a comminutor and is then pumped approximately 450 m to three Imhoff tanks where the wastewater receives primary treatment. In the Imhoff tanks, gross solids settle from the influent. Retention time in the tanks is about six hours, based on a design flow volume of 11,355 m^3/day (3.0 million gallons/day).

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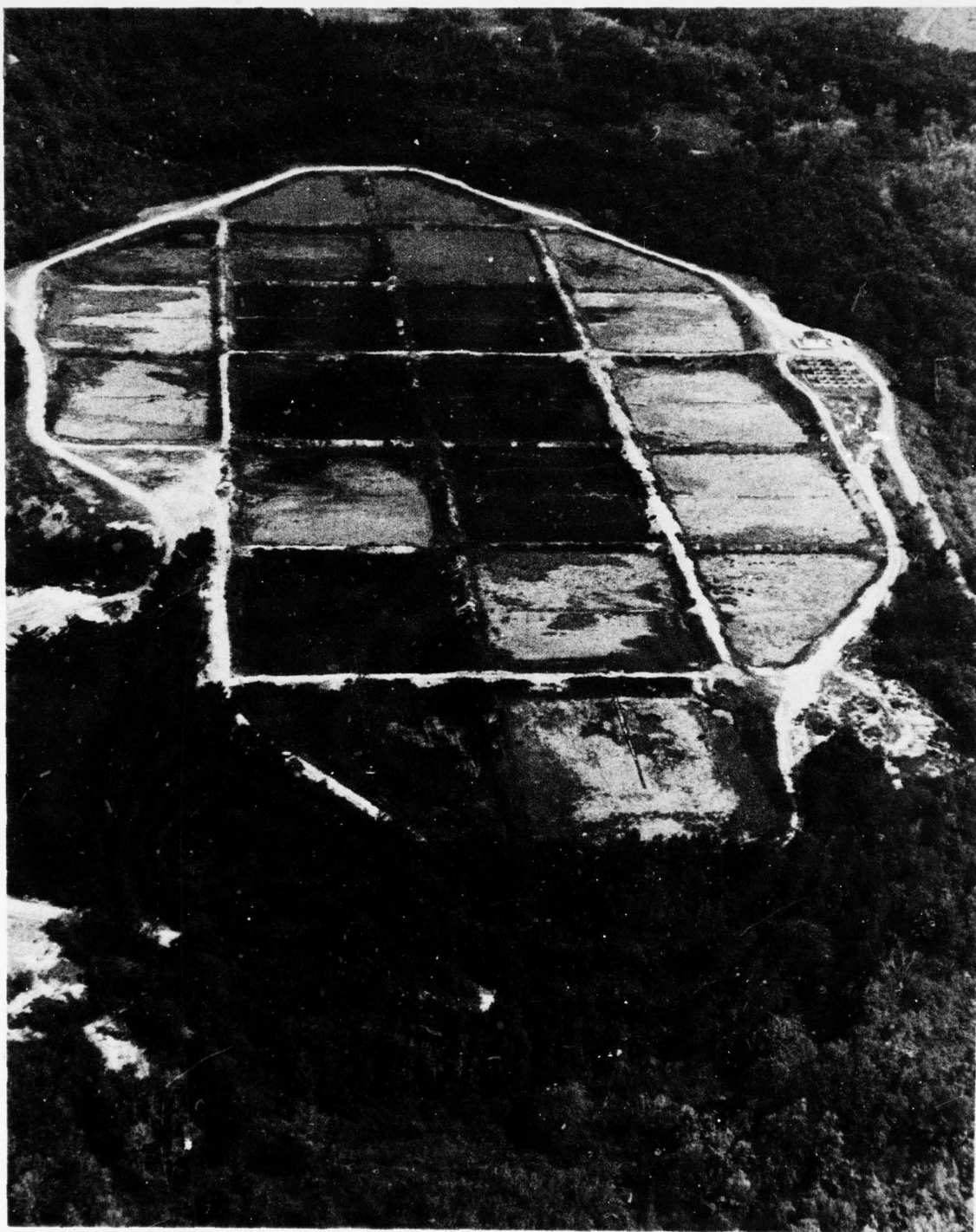
Figure 1. Aerial photograph of sewage treatment facility, Fort Devens, Massachusetts.

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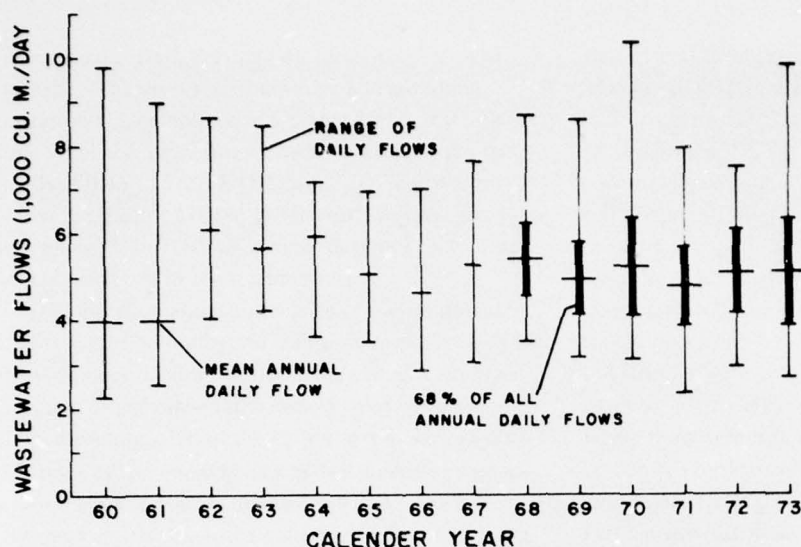
Figure 2. Mean annual daily wastewater flow and annual range of daily wastewater flows (1960–1973).

However, retention varies in relation to influent flow volume and sludge accumulation.

Daily wastewater flows through the treatment facility from 1 January 1960 to 31 December 1973 varied considerably (Fig. 2). Approximately 68% of the daily flows observed from 1 January 1968 to 31 December 1973 was from 4,057 to 5,063 m^3/day. Larger daily flows generally occurred from February to mid-April, while smaller volumes were recorded during August through December (App. A, Fig. A1–A6).

Settleable materials or "sludge" accumulate on the bottom of the Imhoff tanks and are gravimetrically withdrawn to sludge dewatering beds during April and November of each year. These dewatering beds are equipped with underdrains which discharge collected water to the adjacent wetland area.

Unchlorinated effluent from the Imhoff tanks is discharged to 22 treatment beds, each about 0.32 ha. The beds are constructed on a large oval-shaped steep-sided kame, composed of stratified sands, which rises approximately 21 m above the flood plain of the Nashua River. Normally, the treatment beds are used in a rotation sequence, which consists of flooding three beds concurrently with effluent for a two-day period, then allowing a 14-day rest or dry-up period. During holidays and weekends, anticipated effluent flows and precipitation may necessitate using four filter beds rather than the normal three (Fig. 3).

Based upon the application cycle, each treatment bed receives primary sewage effluent approximately 52 days each year. Assuming a mean annual effluent flow of 5,061 m^3/day (1.34 million gallons/day) and equal effluent distribution per unit area, effluent additions to the beds have averaged about 28.3 m/yr (93.8 ft/yr) since 1960.

During the application period, effluent may accumulate on the bed to a thickness of 15 to 50 cm (0.5 to 1.6 ft). All effluent infiltrates the bed within the initial two or three days of the recovery period, thus exposing the bed surface to the atmosphere.

Ice and snow often accumulate on the beds during the winter and can reduce infiltration somewhat. The effluent is sufficiently warm [8 to 12°C (46 to 54°F)] during this time to melt the ice and snow cover and to move into and through the treatment beds.

During the summer, each treatment bed has a good stand of grass vegetation. The major portion of the biomass consists of annual "weedy" grasses; fall panicum (*Panicum dicotomiflorium* Michx.) and barnyard grass (*Echinochloa crusgalli* (L.) Beauv.). Other grass species, as well as *Carex* species, have been observed growing on the beds.

Maintenance procedures prescribed for the treatment beds require that the surface of each treatment bed be excavated periodically to a depth of 0.3 m (1 ft) and that the excavated material be replaced with "bank run" sand and gravel. Prior to replacing the

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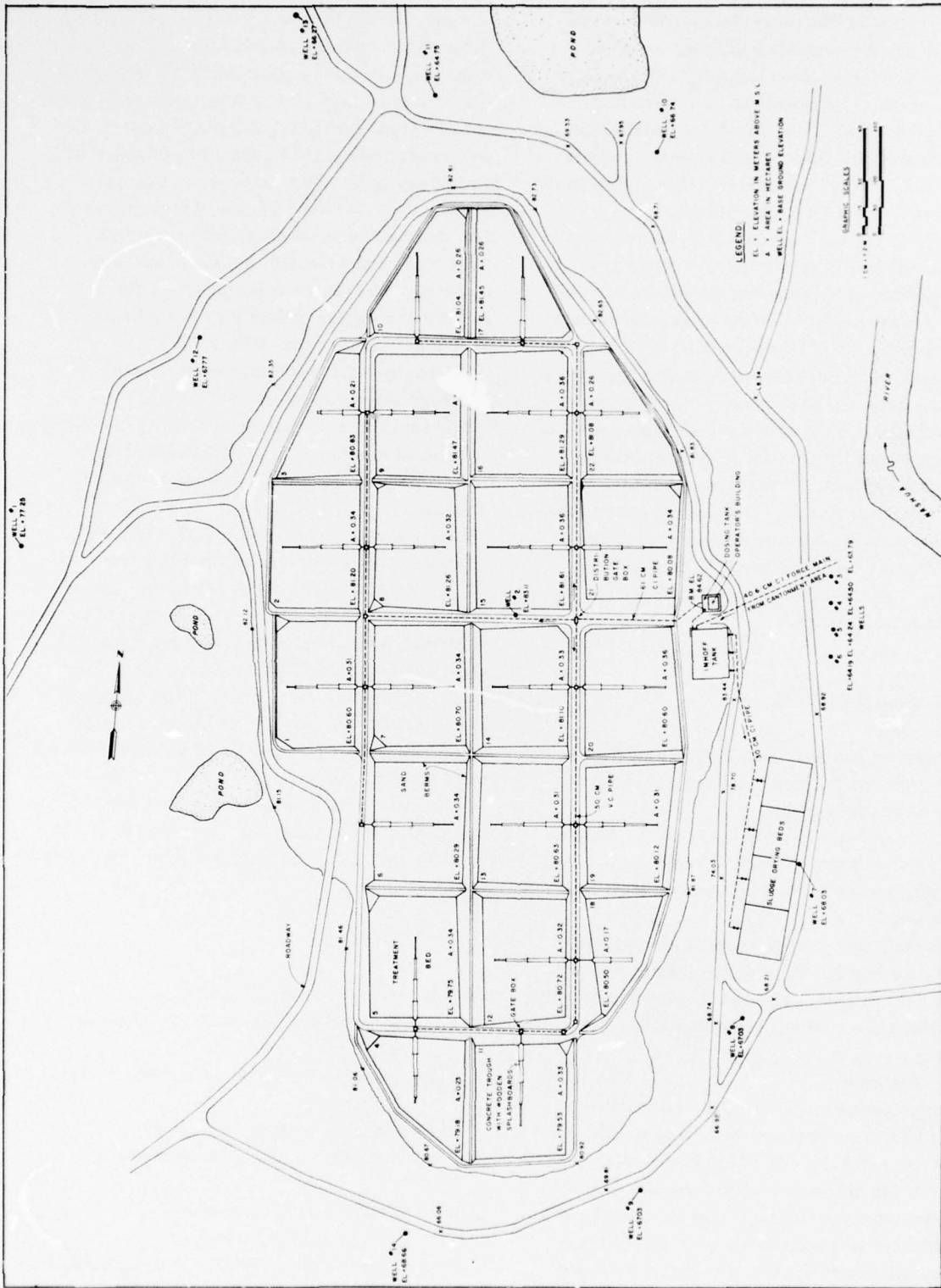
Figure 3. Treatment facility layout plan.

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material excavated, the exposed surface is harrowed or raked. The most recent cleaning and grooming operation was completed in October 1968, at which time 18 of the 22 treatment beds were excavated to a depth of 0.45 to 1.22 m (1.5 to 4.0 ft). Excavation below the specified 0.3-m (1-ft) depth was necessary to remove a "tarlike" layer about 0.45 m (1.5 ft) thick which developed in the treatment beds.

The operation and maintenance of the treatment facility is carried out by two full-time employees. While the treatment facility, including the effluent application area, was designed to handle a given wastewater flow, the application of the daily flows to various combinations of treatment beds was developed from operation experience over an unknown period. The basis for the application cycle is the volume of primary effluent to be treated and the continued capacity of the beds to accept the effluent. Prior emphasis has not been given to the quality of the water resulting beneath the application area; however, as the quality of the water in the Nashua River is improved, the quality of the percolate from the rapid infiltration beds will receive additional consideration.

METHODS AND MATERIALS

The general approach taken in evaluating the impact of primary effluent on treatment beds and underlying groundwater was:

1) to compare the physical and chemical characteristics of soil with those of material of similar texture and origin to which primary effluent had never been applied, and

2) to determine concentrations of selected water quality parameters in regional groundwater, primary effluent, and groundwater beneath and surrounding the land treatment area over a one-year period.

Soil investigations

Continuous dry soil samples were taken from two 6.1-m (20-ft) cores from each of treatment beds 13, 14 15, and 16 (Fig. 3) for physical and chemical determinations. To obtain baseline soil conditions for this study, soil samples were taken from the upper 6.1 m of an adjacent sand and gravel deposit that had never received wastewater effluent. This glacial formation was similar in elevation and origin to the formation on which the treatment beds were constructed. Samples

were also obtained from well 2 (Fig. 3) to a depth of 24.4 m (80.06 ft) for physical determinations. Each boring was made using a pneumatic drill and a 6.3-cm (2.5-in.)-diam, 1.52-m (5.0-ft) sampling spoon. Each core sample was subdivided into horizons differing in soil texture. Coring below a depth of 3.05 m (10 ft) required casing the core hole to prevent sidewall collapse. Each hole was cased with 7.6-cm (3-in.)-diam steel well casing which was pneumatically installed to 3.1 and 4.6-m (10- and 15-ft) depths after soil samples were obtained. Samples were inserted in labeled plastic bags and transported to the laboratory where they were stored at $-10°C$ ($14°F$).

Before physical and chemical analyses, each soil sample was thawed and air-dried at $25°C$ ($77°F$). Particle size distribution was determined by dry sieving with a standard series of nested 20-cm (8-in.) sieves.[6] A roto-tap shaker was used to shake each sample for 5 minutes.

Water retention values at -0.1, -0.3, -1.0, and -15.0-bar tensions were obtained in triplicate for samples derived from two 3.05-m (10-ft) borings and one 24.38-m (80-ft) boring in the application area using the pressure plate and pressure membrane methods.[6] A portion of the air-dried soil sample passing a 2-mm (0.072-in.) sieve was placed within the rubber retaining rings, saturated with tapwater, and allowed to equilibrate under saturated conditions before being subjected to the appropriate pressure.

Saturated hydraulic conductivities were determined using commercially available permeameters (Soil Test model K-620). A constant head was employed. Using Darcy's equation

$$V = Q/AT = Ki$$

where V = velocity of flow (cm/sec)

$\quad A$ = cross-sectional area of the permeameter (cm^2)

$\quad Q$ = quantity of water discharged (ml) in time, T (sec)

$\quad K$ = hydraulic conductivity (cm/sec)

$\quad i$ = hydraulic gradient (dimensionless)

hydraulic conductivities were computed for samples from the 3.05 and 24.28-m borings.

Three different kinds of infiltration tests involving four treatment beds, 13, 14, 19 and 20, were conducted. Metal-ring 38-cm (15-in.)-diam infiltrometers were

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used at three sites in each treatment bed. Infiltration rates were determined by measuring the amount of water entering through the soil surface inside the infiltrometers as a function of time. A head of approximately 5 cm was maintained inside the ring throughout each test. Primary sewage effluent from the treatment plant was used as the infiltration water. Triplicate soil samples were taken at each site and tested for water content determination. After infiltration rate determinations were made on bed 14, this single bed was flooded with all of the effluent flow for a 7-hour period. The infiltration rate was determined as a function of time by measuring the difference between effluent volume increase in the bed and total inflow to the bed. During the flooding test, water level readings were taken at five locations in the bed, every 30-40 min. Because inflow exceeded infiltration, the infiltration rates were influenced by a slow increase in the pressured head.

The third method employed to determine infiltration rate was to observe effluent seepage in bed 1 at the completion of the inundation period. Readings of effluent level were taken at two locations in the bed at approximately 40-min intervals.

Chemical analyses were conducted on samples obtained from two borings in filter beds 13, 14, and 15, which had been stored as described above. Each sample was thawed and air-dried at $25^{\circ}C$, and that portion passing through a sieve with 2-mm openings was evaluated for pH (1:1 ratio of sample to water mixture using a Beckman Zermatic pH meter); organic matter (ignition method);[4] electrical conductivity (1:2 ratio of sample to water mixture);[17] total nitrogen (Kjeldahl method);[4] phosphorus (quinolinium molydophosphate method);[4] orthophosphate (citrate extraction method);[4] total amounts of calcium, magnesium, potassium, manganese, copper, iron and zinc (atomic absorption spectrophotometry of HCl acid extract);[4] boron (Quinalizarian method);[4] sulfur (potassium nitrate-nitric acid digestion method);[5] chlorides (mercuric nitrate method);[4] and cation exchange capacity.[4]

Suction lysimeter investigations

Effluent renovation in relation to increasing soil depth was assessed by collecting percolate samples at five depths in four treatment beds. Samples were obtained using a suction cup apparatus (hereafter called a suction lysimeter), which consisted of $5.0\text{-cm} \times 25\text{-cm}$ soil water sampler (soil moisture no. 1900), rubber

stoppers, 100-ml collection vial, 0.32-cm-ID nylon pressure tubing, 0.32-cm thick-walled rubber tubing and 1.27-cm Hoffman screw clamp. Before installation, distilled water was drawn through each lysimeter and pH was determined to ensure that each porous ceramic cup had a neutral pH, 7.0 ± 0.5.

Triplicate suction lysimeters were placed at 0.31-, 0.61-, 1.52-, 3.05- and 6.1-m depths in treatment beds 13, 14, 15 and 16. Using either a hand-operated bucket auger, or a skid-mounted diamond core drill with a 5.1-cm-diam sampling spoon, the lysimeters were installed on 6.6-m (20-ft) centers. After each lysimeter was placed in position, each core hole was backfilled with clean bank-run coarse- to medium-textured sand. Tapwater was used to compact fill material during backfilling to ensure good surface contact between the lysimeter and the soil.

Percolate sampling was conducted to coincide with the effluent application cycle. Initially, sampling was conducted once every seven weeks but was increased to twice every 7 weeks from April through October 1973. Before sampling, the residual water in each suction lysimeter was removed using a hand-operated suction pump. A negative pressure, –0.8 to –0.9 bar, was applied and maintained on each lysimeter for 2 hours. Samples from triplicate suction lysimeters positioned at the same depths and in the same treatment beds were composited for laboratory analysis of the following parameters using chemical procedures outlined in Standard Methods:[2] pH (glass electrode method), electrical conductivity (Wheatstone bridge), chemical oxygen demand; organic Kjeldahl nitrogen, NH_4-N (Nesslerization); nitrite nitrogen (colorimetric method), total and orthophosphate potassium (stannous chloride method); and sulfate turbidimetric method. Nitrate nitrogen[14] and chlorides were determined using specific ion electrodes.[21]

Groundwater and effluent investigations

The quality of groundwater below the treatment beds and in the area peripheral to the application site was monitored by collecting and analyzing water samples from 14 groundwater observation wells. Six wells (3-8), consisting of 3.2-cm (1.25-in.)-diam metal riser pipe and 0.91-m (3.0-ft) no. 10 metal well screens, were installed during October 1972. Six additional wells (9-14), consisting of PVC riser pipe and 3.05-m (10-ft) no. 20 PVC well screens, were added during April 1973 (Fig. 3). Each well screen was positioned

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so that the upper portion of the water table could be sampled throughout the hydrologic year. Depths to water table in wells no. 1 and no. 2 were too great for suction pumping; therefore, 7.6-cm (3-in.) and 15.2-cm (6-in.)-diam metal riser pipe and 1.52-m (5-ft) well screens were installed in October 1972.

All wells were installed by wash drilling, using either skid-mounted or truck-mounted diamond core drills. All holes were cased during the drilling operations with NW and SW size flush joint casings. Tapwater was used in the installation of casing for removing substrate from the casing during the drilling operation to prevent possible contamination. Following installation, each well was pumped for 30 min to remove any fine sediments present in the well.

Groundwater samples were collected biweekly from each observation well. The large-diameter deep wells were sampled using a 400-ml (0.11-gal) Kemmerer water sampler. Samples from the shallow 3.2-cm-diam wells were collected directly into sterile, 1,000-ml (0.26-gal) labeled fleakers using a hand-operated suction pump.

A composite sample of primary effluent was collected during the 24-hour period preceding each biweekly well sampling. The composite sample was collected with a proportional sampler equipped with a refrigeration unit which cooled the effulent to 5°C. Following thorough mixing of 18 liters of composited effluent, a 1,000-ml (0.26-gal) subsample was drawn off for laboratory analysis.

Groundwater and composite effluent samples were transported to the laboratory within 4 hours after collection for immediate chemical and biological analyses. All samples were analyzed for the following parameters according to procedures outlined in Standard Methods for Examination of Water and Wastewaters:[2] pH (glass electrode method); electrical conductivity (Wheatstone bridge); alkalinity (potentiometric method); five-day biochemical oxygen demand (Winkler determination); organic Kjeldahl nitrogen (potentiometric titration); ammonium nitrogen (Nesslerization); nitrite nitrogen (colorimetric method); total and orthophosphates (stannous chloride method); sulfate (turbidimetric method); and total coliform bacteria (membrane filter technique). Hardness,[20] nitrate nitrogen,[14] and chlorides[21] were all determined utilizing specific ion electrodes.

RESULTS AND DISCUSSION

Soil investigations

Physical determinations. A stratigraphic section (Fig. 4) constructued from the drillers' boring logs, depicts the various horizons in the upper 6.1 m (20 ft) of the filter beds (A–A'). The formation in which the treatment beds were constructed is primarily poorly graded sands or gravelly sands, with interspersed lenses of silty sand and sandy gravels. The low-lying areas adjacent to the beds consist of medium- to fine-textured sands with some lenses of gravel.

Sieve analysis of the substrate samples from the treatment bed formation revealed that most strata contained only a small percentage (1 to 3%) of particles finer than 0.053 mm. Only 7 of the 39 treatment bed strata samples contained more than 5% "fines" (silt and clay). The tabular data (App. A, Table AI) suggest appreciable differences in distribution of particle size between the various horizons in the beds. About one-half of the strata sampled in beds 14, 15, and 16 was composed of over 40% of particles falling into the gravel-size range (2 mm). The percentage of particles greater than 2 mm was inadvertently not determined on samples from bed 13.

Background samples were collected from a formation consisting of stratified horizons of sand and gravels. Coarse to medium sand comprised the major portion of the 3.05-m (10-ft) profile, with gravel forming an important percentage in the 0.3 to 0.9-m (1 to 3-ft) horizon. Below the 0.9-m depth, the formation consisted of 2 m (6 ft) of medium sand, underlain by strata of sandy gravel and gravelly sand, similar to the material encountered in the well no. 2 profile. Silts and clays composed about 10 to 15% of the volume in the upper profile but less than 1% below the 0.5-m (1.6-ft) depth.

Water contents retained by boring samples were similar and representative of unconsolidated coarse-textured material, with low percentages of silt- and clay-sized particles (Fig. 5). The water retained in the samples decreased rapidly from saturation to the –1.0-bar tension where water contents were generally 1 to 2%. Moisture content changed very little between –1.0 bar and –15.0 bar tensions. Water retained in each sample at the –15 bar tension was less than 1.0%. Moisture retention was influenced more
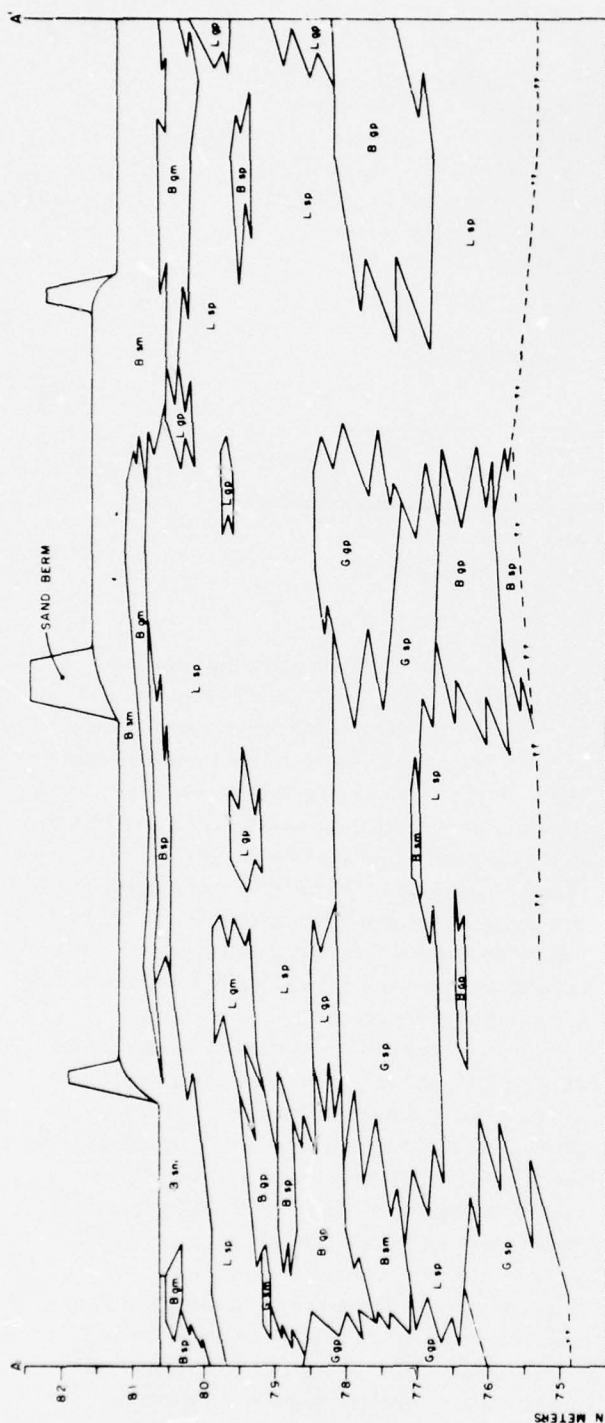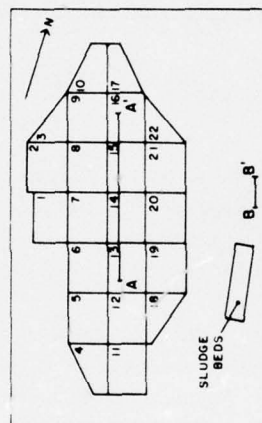
STRATIGRAPHIC SECTIONS ALONG A-A' AND B-B'
SEWAGE TREATMENT FACILITIES
FORT DEVENS, MASSACHUSETTS

LOCATION MAP

ELEVATION IN METERS
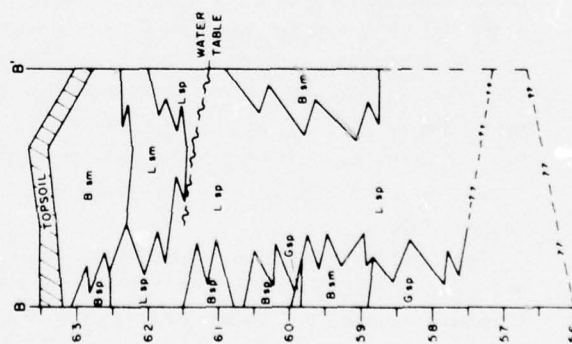
Figure 4. Stratigraphic section of treatment plant formation.

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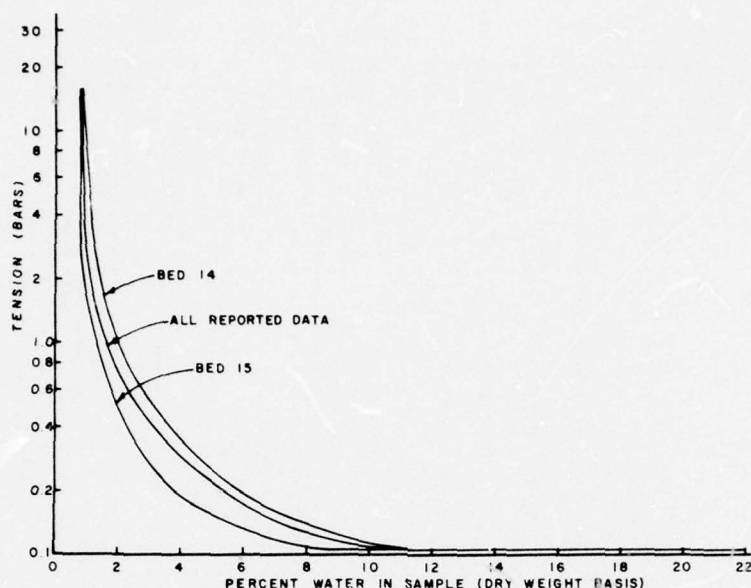
Figure 5. Moisture release curves of soil samples.

by the percentages of silt and clay than by the percentages of the larger grain-sized particles. Samples with 4% fines held more moisture at the –0.1 and –0.3 bar tensions than samples with lesser amounts of silt and clay.

During an effluent flooding cycle, the surface soil approaches saturation, and then drains to field capacity during the recovery period. Assuming that bulk density is 1.2 g/cm^3 and that a tension of –0.3 bars approximates field capacity, the average depth of water held in various horizons is about 0.5 mm/cm soil depth. This means that, following the rest period, there is sufficient void space in about 3 m (10 ft) of soil to hold all the effluent applied during a 3-day application period to three treatment beds. Since water held in the soil at tensions less than about –0.3 bars moves through the sand and gravel horizons, the treatment beds are "dewatered" during the recovery period as water percolates through the treatment-bed formation, eventually adding to groundwater.

Hydraulic conductivities of samples from well no. 2 and treatment bed 15 that had passed through the 2-mm sieve were measured with a saturated column under a constant head. Permeabilities were of the order 10^{-3} to 10^{-2} cm/sec (4 × 10^{-4} to 4 ×10^{-3} in./sec) (Table I). Most permeabilities could be classified as rapid to very rapid with only a few moderately permeable horizons.

Moderately permeable strata were generally those with greater than 4% "fines" and 10% very fine sand. Rapid or very rapid permeabilities were generally observed in material with less than 10% particles smaller than 0.10 mm. Theoretically, each horizon easily transmits the designed daily wastewater flow of 11,355 m^3/day (3 mgd) provided a 1-hectare (2.47-acre) application area is used. The occurrence of effluent ponding on the treatment bed surfaces shows that the infiltration rate decreases during the application period, thus limiting the rate at which effluent moves into and through the formation.

Infiltration tests were conducted on treatment beds (1, 13, 14, 19, and 20), which represented a wide variety of soil moisture conditions and bed surface conditions. Soil moisture ranged from near saturation to very low water content, and surface conditions consisted of an organic mat overlying the soil surface or an exposed bare soil. When time infiltration tests were performed, the following conditions existed: the 2-day flooding cycle had just been completed for bed 1 and it was under about 50 cm of effluent; the 14-day rest period of beds 13 and 14 had just been completed, representing minimum moisture in the surface soil; and beds 19 and 20 were about 30 hours into their rest period, representing a condition where surface soil of these beds was free from standing water but still

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Table I. Hydraulic conductivities of substrate samples from infiltration area.

Sample depth (m)	Hydraulic conductivity (cm/sec) $\times 10^{-2}$
Filter bed 15	
0.0 - 0.15	1.14
0.15- 0.46	0.80
0.46- 1.22	1.75
1.22- 1.52	1.90
1.52- 3.05	0.14
3.05- 4.27	0.46
4.27- 4.57	0.42
4.57- 4.88	1.78
4.88- 5.79	0.15
5.79- 6.10	0.20
Well no. 2	
0.0 - 0.24	0.28
0.24- 1.28	0.19
1.28- 1.52	0.20
1.52- 2.71	0.80
2.71- 3.05	0.17
3.05- 4.57	0.14
6.10- 7.62	0.83
7.62- 9.14	0.03
9.14-10.67	1.70
10.67-12.19	0.39
12.19-13.72	1.00
13.72-15.24	0.60
15.24-16.74	0.70
16.74-18.29	0.21
18.29-19.81	0.23
19.81-21.34	2.40
22.86-24.38	0.10

quite wet compared with surface soil of beds 13 and 14. All beds except bed 19 had been "cleaned" in 1968.

Because effluent intake rate depends on moisture content, the water contents of the surface soils were measured at each site where infiltrometers were used. Table II summarizes the results of the metal-ring infiltrometer tests in beds 13, 14, 19, and 20. These data show the effect of initial soil moisture content and time on effluent intake rate. Treatment beds with the slowest infiltration rates were those having the highest initial soil moisture content levels.

Measured soil moisture contents of the surface 0.0 to 0.15 m (0 to 6 in.) in beds 13 and 14 were greater than water retention data would suggest, probably because of the high organic matter incorporated in these soil samples. Bed 13, having the lowest initial water content, also had the greatest effluent intake rate.

Table III and Figure 6 show the effect of time on effluent intake rate in bed 14. Because all effluent flow during a 7-hour period was directed to bed 14, the intake rates do not represent rates under actual management practices where effluent is simultaneously applied on three beds and pressure head buildup is slower. Under normal operating conditions, it takes longer than 6 hours for the intake rate to approach 2.2 cm/hr or the time that ponding starts.

Ponding on treatment beds begins when effluent flow from the treatment plant exceeds the intake rate of the treatment beds. The seepage test conducted in bed 1 suggests that, at the completion of the inundation cycle following actual management practices, effluent intake rate had decreased to about 1.5 cm/hr or lower (Table IV). Note that during flooding and seepage tests and during normal flooding of the beds the intake rate is increased because of the increased pressure head that develops as effluent ponds on the treatment beds. Data in Table IV suggest that the intake rate could decrease to about 1 cm/hr or lower at the end of a flooding cycle when the pressure head has diminished to near zero.

Several factors are responsible for decreasing intake rates with time. The most obvious reason for the decline is that soils become water filled, causing increased frictional resistance to waterflow. Thus, some reduction in intake rate with time is no doubt due to physical and biological clogging of the soil.

Visual observation of approximately 48 corings made to depths of 0.3, 3.1, and 6.2 m (1, 10, and 20 ft) in treatment beds 13, 14, 15, and 16 failed to indicate the formation of a horizon capable of limiting vertical movement of water. (Observations made in treatment bed 19, one of four beds not excavated in 1968, revealed the presence of a very definite "black tarlike" horizon which had a tainted sulfide and petroleum odor. This horizon was approximately 0.45 m (1.5 ft) thick and had formed at a depth of about 0.45 to 0.60 m (1.5-2.0 ft). Formation of this horizon most probably required many years.)

Before 1968, the periodic "grooming" of the beds included the removal and replacement of the upper

Table II. Initial and average effluent intake rates and water contents (dry weight basis) of four treatment beds.

Bed	Site 1	Site 2	Site 3	Average
Bed 13				
% H_2O	2.4	3.8	3.1	3.1
0-3 min IR (cm/hr)*	143	117	143	134
3-60 min IR (cm/hr)†	49	35	33	39
Bed 14				
% H_2O	6.2	6.7	14.5	9.1
0-3 min IR (cm/hr)	107	97	10.2	71.4
3-60 min IR (cm/hr)	12.2	40.4	2.6	18.4
Bed 19				
% H_2O	18.0	9.7	15.5	14.4
0-3 min IR (cm/hr)	15.3	15.3	15.3	15.3
3-60 min IR (cm/hr)	3.8	4.9	3.4	4.0
Bed 20				
% H_2O	12.0	12.7	12.8	12.5
0-3 min IR (cm/hr)	12.7	14.2	5.3	10.7
3-60 min IR (cm/hr)	5.7	4.2	6.0	5.3

* Initial intake rate.

† Average intake rate.

Table III. Effluent intake rate of treatment bed 14 determined by flooding test.*

Elapsed time (hr)	Intake rate (cm/hr)
2.2	6.0
2.8	5.8
3.3	4.3
3.9	2.5
4.6	2.5
5.3	2.4
6.0	2.1

* Two hours were required to completely inundate the soil surface of this treatment bed.

Table IV. Effluent intake rate of treatment bed 1 following 2-day inundation.*

Elasped time (hr)	Intake rate (cm/hr)
0.8	1.53
1.8	1.47
2.9	1.37
Average	1.46

* Intake rates are average values for the corresponding elapsed time since measurements were started.

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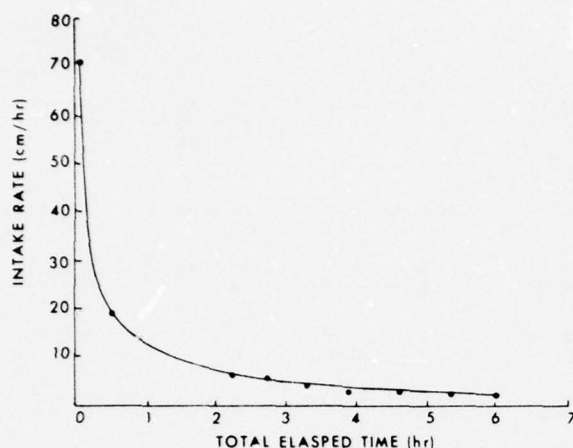
Figure 6. Effluent intake rate of treatment bed 14 as a function of time.

0.3 m (1 ft) of sand and gravel material, a practice that would have permitted the black horizon to form. During this time, there were numerous reports of crankcase oil and kitchen grease flowing through the Imhoff tanks to the treatment beds. In 1968, although the 2-day inundation followed by 14-day recovery application cycle was being practiced, the movement of effluent into the beds was reportedly negligible. Water was standing on all beds at the conclusion of the recovery period. The black layer was reportedly the primary cause of the standing water.

Following the excavation of 18 treatment beds to a depth below this horizon, the infiltration rate was restored. Infiltration of the effluent had apparently recovered in those beds not excavated in 1968. Since the black horizon did not form a continuous stratum beneath the entire infiltration area in these beds, it is possible that the black layer had been decaying since 1968, or that the reduced infiltration rate was related to other factors stemming from the continuous inundation of the treatment bed. Correcting the dumping of oil and grease into sewerlines and taking better care of grease traps had undoubtedly prevented the flow of oil and grease to the treatment beds since 1968.

Chemical determinations. Data from the chemical analyses of the treatment beds and background samples are presented in Table V. Approximations of chemical characteristics of various horizons in the treatment beds before primary effluent was applied after either the construction or the cleaning and grooming operation are

best shown by comparing data for background samples collected at depths greater than 0.6 m. From this portion of the background fromation, "bank run" sand and gravel replaced the excavated material removed during the cleaning operation. Chemical analysis of the soil samples collected in the treatment beds showed that the beds were acidic, pH 5.6 to 6.4 throughout the 3.05-m (10-ft) profile. However, the treatment beds were less acidic than the background samples (pH of 4.4 to 5.2). Organic matter in the treatment beds was substantially higher than the 0.4% observed in the background soil samples. Surface horizons in the treatment beds were 1.2 to 1.4%. Organic matter levels decreased to about 0.7% at the 3-m (1-ft) depth. Organic matter accumulation within the upper 0.15 m (0.5 ft) of the treatment beds profile was about 0.2%/yr, since the treatment beds were "cleaned and groomed" in 1968.

The electrical conductivities of treatment bed samples were 0.06 to 0.23 μmhos, with the highest values observed in samples from the surface horizons. The conductivities of background samples were less than the detection limit below the 0.15-m (0.5-ft) depth.

Cation exchange capacities of background soil samples varied from 2.0 to 8.0 meq/100 g in the surface horizons and were about 0.5 meq/100 g at depths greater than 0.5 m (1.6 ft). The higher exchange capacities in the surface horizons of the background formation appeared to be directly related to the level of organic matter and the greater percentage of silt- and clay-size particles. Cation exchange capacities of soil samples taken in treatment beds were relatively constant throughout the 3-m sampling depth and were 0.79 to 1.5 meq/100 g. Compared with background samples from lower strata, the exchange capacity in the treatment beds increased. The exchange capacity in the treatment beds had apparently increased during the 6 years since the beds were last renovated. It is uncertain at this time what factors caused the increase, but the accumulation of organic matter above those observed for background samples was probably an important factor.

Phosphorus concentrations in soils from treatment bed profiles were 5,570 to 7,060 kg/ha-m (1,515 to 1,920 lb/ac-ft), representing an increase of 250% over observed background levels. Phosphorus levels in the surface horizons of treatment beds were slightly greater than levels in lower horizons. Even so, phosphorus levels were somewhat constant in the 3-m (10-ft) bed profile. Approximately 60% of the total phosphorus

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Table V. Chemical characteristics of treatment bed and background samples.

Sample depth (m)	pH	Organic matter (%)	Electrical conductivity (µmhos/cm)	Nutrient (ppm)													CEC* (meq/100 g)
				N	P	Ca	Mg	K	Mn	Na	Cu	Fe	Zn	B	S	Cl	
Filter Bed 13†																	
0.0 -0.15	5.7	1.23	0.23	460	472	488	712	365	36	21	29	3,470	31	0.8	2.7	15	1.5
0.15-0.46	5.7	0.79	0.18	267	472	378	923	357	42	23	29	4,143	26	0.6	1.8	14	1.0
0.46-0.61	5.8	0.76	0.18	252	550	350	1,008	405	54	24	32	4,340	25	0.8	2.0	13	1.0
0.61-0.91	5.8	0.61	0.15	150	397	295	1,075	405	52	22	16	4,480	22	0.8	2.0	12	0.8
0.91-1.52	5.8	0.75	0.16	132	541	591	2,278	772	78	27	46	7,045	41	0.8	3.0	13	1.3
1.52-1.98	6.3	0.54	0.08	113	359	467	1,227	590	55	23	22	4,813	25	0.8	2.3	13	0.8
1.98-2.59	6.2	0.59	0.10	92	414	588	2,010	935	82	26	25	6,905	37	0.8	2.7	11	1.1
2.59-3.05	6.2	0.76	0.12	105	552	958	2,640	1,520	107	26	25	11,375	39	0.9	3.0	11	1.3
Filter Bed 14†																	
0.0 -0.30	5.6	1.13	0.17	417	571	1,020	863	703	83	23	18	8,385	37	3.4	2.0	5	1.4
0.30-0.61	5.8	1.00	0.15	382	623	722	625	625	78	25	14	6,750	32	2.5	2.1	8	1.4
0.61-1.22	6.0	0.88	0.11	215	599	600	664	847	94	21	12	7,124	30	2.4	2.0	5	1.2
1.22-1.52	6.2	0.90	0.08	684	537	726	1,133	1,100	106	23	16	8,250	43	2.0	1.6	6	1.5
1.52-2.74	6.3	0.62	0.06	131	434	706	830	1,082	86	25	14	6,858	62	2.0	2.0	6	1.0
2.74-3.05	6.4	0.72	0.07	126	529	1,369	1,440	1,652	1,162	25	18	11,188	42	1.8	2.0	5	1.2
Filter Bed 15†																	
0.0 -0.15	5.6	1.39	0.21	531	691	970	2,002	528	73	22	19	7,435	43	1.1	3.1	13	1.4
0.15-0.46	6.0	1.08	0.13	283	796	819	1,594	695	84	14	14	3,750	37	0.9	3.1	7	1.4
0.46-1.22	5.9	0.93	0.16	231	669	552	1,524	806	80	16	11	6,029	28	0.8	3.1	6	1.3
1.22-1.52	6.1	0.71	0.10	112	412	482	1,688	834	83	19	9	6,158	23	0.3	3.3	6	1.1
1.52-3.05	6.2	0.55	0.09	82	368	480	1,516	784	80	10	17	6,188	22	0.4	3.3	5	0.8
Background Samples																	
0.0 -0.15	4.4	5.41	0.03	920	299	108	484	188	79	5.3	4.0	8,469	13	4.5	1.3	10	8.8
0.15-0.30	4.6	2.36	0.0	270	301	66	781	200	60	4.5	4.0	10,438	12	3.8	1.2	6	2.4
0.30-0.46	4.6	1.77	0.0	210	303	60	1,229	275	73	4.0	6.0	9,188	13	5.3	0.6	4	2.3
0.46-0.61	5.0	0.45	0.0	80	153	59	760	344	69	3.3	4.0	3,688	8	4.1	0.4	4	1.0
0.61-0.91	4.8	0.43	0.0	60	160	78	693	300	48	3.3	4.0	3,500	7	4.1	1.5	4	0.9
0.91-1.83	5.2	0.26	0.0	35	146	132	718	425	66	3.2	4.5	3,094	8	0.8	0.6		0.6
1.83-2.44	5.2	0.39	0.0	50	228	186	896	437	75	2.9	5.0	4,500	9	1.9	0.4	2	0.8
2.44-3.05	5.2	0.40	0.0	60	304	321	729	519	62	2.4	3.5	4,406	10	6.2	0.6	2	0.9

* Cation exchange capacity.

† Average of data from two cores.

observed in the treatment beds was orthophosphate as compared with 10 to 20% in background samples. The form of remaining phosphorus was not ascertained but is believed to have been primarily phosphate precipitates of calcium, magnesium and iron. In bed 13, there appeared to be a direct relationship between the concentrations of phosphorus and the concentrations of calcium, magnesium and iron. This was not as apparent for other treatment beds.

Calcium, magnesium, potassium and iron levels varied appreciably between different horizons and between different beds. In general, these cations increased in concentration with increasing depth. Compared with background sample levels, calcium, magnesium, potassium and iron in the treatment beds were, respectively, 300 to 500%, 100 to 270%, 200 to 270%, and 140 to 175% greater.

Sodium, copper and zinc concentrations varied in the treatment beds but were generally more uniform within the profile of each bed. Sodium ranged from 10 to 27 ppm, zinc from 22 to 62 ppm, and copper from 9 to 46 ppm. Treatment bed samples contained three to six times higher concentrations of these elements than did the background samples.

Chloride (5 to 15 ppm) and sulfur (1.6 to 3.3 ppm) levels in the treatment beds were about 200% and 400%, respectively, greater than background levels. Concentrations of chloride and sulfur did not vary appreciably, however, within the treatment bed profiles. Chloride levels in bed 13 averaged about 8 ppm higher than background levels, while chloride levels in beds 14 and 15 were 1 to 3 ppm higher. Sulfur levels were 1 to 2 ppm higher than background levels.

Boron concentrations were 0.3 to 3.4 ppm in the treatment bed samples; however, these concentrations were about one-third the concentrations observed for background sand and gravel samples. Reduction in boron levels was due to the solubility of this ion, which would have permitted it to be carried through the treatment beds in the percolating water.[7]

Analysis of samples taken from the treatment beds showed that total nitrogen was highest, 750 to 950 kg/ha, in the surface horizon of each bed. Nitrogen levels decreased with increasing depth in treatment beds to approximately 180 kg/ha at the 3.05-m depth. Compared with nitrogen concentration in background samples, 72 to 90 kg/ha, total nitrogen in the surface horizons increased eight- to tenfold since the beds were excavated. Calculations show nitrogen accumulation

averaged about 680 kg/ha yr (670 lb/ac yr) in the surface meter of soil since the 1968 "cleaning and grooming" operation.

Table VI summarizes the chemical changes in the rapid infiltration beds after having received primary sewage effluent for 30 years. In comparison with the background soil samples, the treatment beds had substantially higher levels of most chemicals. Surface horizons of the treatment beds which had received primary effluent for only 5 years, had significant levels of organic matter, nitrogen, phosphorus, and other cations. Only boron appeared to have decreased from observed background levels.

Table VI. Summary of soil chemical changes in treatment beds compared with those in background soils.

Parameter	Change
pH	Less acidic
Organic matter	Increased
Electrical conductivity	Higher
Total nitrogen	Higher
Total phosphorus	+250%
Calcium	+300–500%
Magnesium	+100–270%
Potassium	+200–270%
Iron	+140–175%
Sodium	Higher
Copper, zinc	Higher
Boron	Decrease
Chloride, sulfur	Higher
Cation exchange capacity	Slight increase

Suction lysimeter investigations

Suction lysimeters installed in October 1972 produced percolate samples from all five sampling depths in the four treatment beds. The 60 suction lysimeters were operable before the cold periods; however, attempts to collect samples during December through early April were not successful because of water freezing in the small-diameter tubing. Sampling during warmer periods was more encouraging; however, sample volumes from deeper suction lysimeters were often too small for conducting laboratory analyses.

To draw water into suction lysimeters, soil water must be present at tensions less than the negative

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pressures applied here, –0.8 to –0.9 bar. Soil water held at field capacity in the coarse-textured sand and gravel strata was about 3 to 4%, half of which could be drawn into the suction lysimeters before inundating the treatment beds. To obtain adequate sample volume for chemical analysis during the 2-hour sampling period, strata must be approaching saturated conditions. Since suction lysimeters were located some 12 m above the water table, percolate sampling was conducted after 24 hours of effluent application. Even so, this procedure produced composited percolate samples totaling less than 100 ml, particularly sampling from suction lysimeters at the 3.0 and 6.1-m depths. Sampling after 48 hours of effluent application did not appreciably increase sample volumes from suction lysimeters at 3.0- and 6.2-m depths.

Calculating downward movement of percolate water through the treatment beds showed that percolate would have moved 1.0 to 1.5 m after 24 hours and would not have reached the 3-m depth until sometime after the 48 hours of effluent application. Thus, extracting portions of the percolating water with suction lysimeters must coincide with the probable arrival of the lysimeters at the sampling depths. Since all lysimeters were sampled at the same time in relation to initiating effluent application, small sample volumes were generally collected from suction lysimeters at depths greater than 1.5 m.

Analytical results for percolate samples obtained from suction lysimeters installed at the 0.3-, 0.9-, and 1.5-m (1-, 3-, and 5-ft) depths are summarized in Table VII. Data for suction lysimeters at these depths showed considerable variation in the concentrations of most parameters; however, substantial variations in the concentration of effluent parameters accounted for much of the observed fluctuation. Mean values of chemical oxygen demand (COD) in suction lysimeter samples ranged from 53 to 92 mg/l at the 0.3-m depth and from 67 to 83 mg/l at the 1.5-m depth. Compared with the mean COD level of the effluent, 208 mg/l, COD levels in percolate samples were 56-75% less.

Total nitrogen in the suction lysimeter samples averaged about 35 mg/l or about 30% less than effluent levels. Although total nitrogen levels were comparable for the three sampling depths, differences were noted in the concentrations of organic-N, NH_4-N, and NO_3-N. Concentrations of NO_2-N ranged from 0.002 to 1.2 mg/l in the percolate, which were generally higher than effluent levels, 0.002 to 0.1 mg/l.

Organic and NH_4-N concentrations in the percolate were approximately equal, 12 to 15 mg/l, which were about 40 to 50% less than levels observed in the effluent, 25 mg/l and 24 mg/l. Although organic-N and NH_4-N levels in the percolate varied substantially, concentrations appeared to decrease slightly with increasing sampling depth.

NO_3-N concentrations in the percolate were greater than levels found in the effluent. NO_3-N at the 0.3-m depth averaged 2.5 and 9.4 mg/l in beds 13 and 14 and 15 and 16, respectively. Samples from the 1.5-m depth averaged 11 mg/l in beds 13 and 14 and 6 mg/l in samples from beds 15 and 16.

Lower organic-N and NH_4-N levels and higher NO_2-N and NO_3-N concentrations in the suction lysimeter samples indicate nitrification of nitrogenous compounds in the surface horizons of the treatment beds. Reduction of total nitrogen by about 30% probably reflects nitrogen losses resulting from the physical removal of nitrogenous organic compounds and the sorption of NH_4-N by the soil exchange complex. The period between initiating effluent application and suction lysimeter sampling probably was not long enough for conditions suitable for denitrification to develop.

Total and orthophosphorus levels in the percolate samples generally ranged from 4 to 16 mg/l with mean values of 7 to 10 mg/l (Table VII). Compared with effluent PO_4-P concentrations, percolate levels were generally 3 mg/l less. Phosphorus reduction appeared to have occurred in the surface 0.3-m horizon with little additional decrease within the 0.3 to 1.5-m depth. This observation, together with the PO_4-P concentrations in soil samples, suggests that the surface horizons of the treatment beds may have been nearing the PO_4-P capacity.

Sulfate concentrations in the percolate, 31 to 42 mg/l, were generally less than effluent levels, 45 mg/l. Although sulfate levels in the percolate were less than effluent concentrations, soil chemical analyses showed that sulfates did not accumulate substantially in the treatment bed profiles. Sulfates may accumulate in bed surfaces possibly as organic sulfate compounds or as precipitates for a short time, but are finally released, either as gases to the atmosphere, or in their anionic forms to the percolating water, and subsequently carried through the soil profile.

Chloride levels in the percolate samples were comparable to effluent levels, although some higher values

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Table VII. Chemical characteristics of suction lysimeter samples from three depths in four treatment beds and of primary sewage effluent.

| | Effluent | | 0.3-m Sampling depth | | | | 0.9-m Sampling depth | | | | 1.5-m Sampling depth | | | |
| | | | Beds 13 and 14 | | Beds 15 and 16 | | Beds 13 and 14 | | Beds 15 and 16 | | Beds 13 and 14 | | Beds 15 and 16 | |
Parameter*	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range	Mean	Range
pH (standard units)	7.0	6.7-7.8	7.3	6.8-7.8	7.3	6.7-7.9	7.3	6.6-7.9	7.4	6.8-7.9	7.0	5.9-7.9	7.3	6.3-7.9
Conductivity (μmhos)	557	441-750	514	350-820	465	400-730	516	365-710	456	360-610	503	270-825	472	367-568
COD	208	79-300	92	60-150	53	10-113	84	50-185	65	10-128	67	25-130	83	20-210
Total Nitrogen	50	32-79	34	21-56	34	15-72	33	18-57	35	15-55	34	4-68	31	5-73
Organic-N	25	18-36	16	11-24	11	3-20	15	9-22	15	5.0-22	12	2-26	13	2.0-28
NH_4-N	24	13-42	15	10-25	13	11-22	13	8-18	18	8.9-30	11	2-22	12	2.0-19
NO_3-N	1.1	0.02-2.5	2.5	0.8-6.0	9.4	0.7-30	5.3	0.7-17	2.0	0.8-2.4	11	0.7-20	6	1-25
NO_2-N	0.02	0.002-0.1	0.3	0.05-1.2	0.10	0.03-0.5	0.04	0.03-0.08	0.09	0.01-0.8	0.08	0.002-0.2	0.1	0.05-0.2
Total PO_4-P	11	6-16	9	6-13	8	4-13	9	6-11	10	6-14	9	5-15	9	5-15
Ortho PO_4-P	10	4-14	8	5-13	7	3-13	8	5-11	10	5-14	8	5-12	8	4-14
Sulfate	45	28-55	36	18-51	50	30-80	31	8-54	32	18-44	32	20-48	42	17-66
Chloride	157	105-190	137	74-190	223	105-365	161	80-220	143	55-195	137	90-245	198	96-340
Iron	0.2	0.01-0.5	0.2	0.03-0.5	0.05	0.49-0.1	0.1	0.02-0.2	0.1	0.08-0.2	0.08	0.02-0.2	0.1	0.02-0.3

* In mg/l unless otherwise indicated.

17
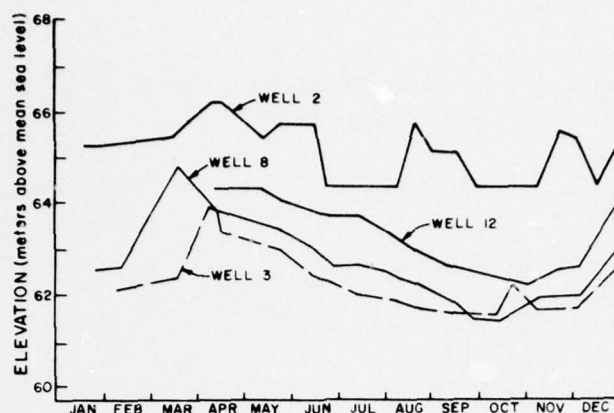
Figure 7. Water table elevations in selected observation wells (1973).

Table VIII. Water additions and losses to treatment beds (1973).*

Month	Precipitation (cm)	Potential evaporation (cm)	Effluent additions (cm)	Total water inputs (cm)	Total water inputs × 100% effluent additions (%)
January	10.9	1.8	260	269.1	103.5
February	7.2	1.9	278	283.3	101.9
March	7.5	3.1	261	265.4	101.7
April	15.0	5.4	302	311.6	103.2
May	10.6	8.6	266	268.0	100.8
June	8.1	9.4	242	240.7	99.5
July	4.3	10.1	221	215.2	97.6
August	6.7	9.0	196	193.7	98.8
September	5.0	6.0	177	176	99.4
October	8.2	3.8	161	165.4	102.7
November	5.7	4.1	158	159.6	101.0
December	22.4	2.8	185	204.6	110.6
Total	111.6	66.0	2,707	2,752.6	$\bar{X} = +1.7$

* Potential evaporation estimated at 66 cm/yr.[26]

were observed in the percolate samples. High values suggest a slight buildup of chlorides in the surface strata, followed by leaching through the profiles of the treatment beds during the next inundation period.

Renovation of effluent during the initial portion of the inundation period appears to have occurred within the surface 0.3 m of the treatment beds. In this horizon, COD, PO_4-P, and total nitrogen levels in the effluent were reduced. Further, in this horizon, transformation of nitrogenous compounds to NO_3-N apparently was occurring.

Groundwater and effluent investigations

The volume of primary sewage effluent applied to the treatment beds in 1973 varied from 2,676 to 9,841 m^3/day (0.7 to 2.6 mgd) with an average daily volume of 5,049 m^3/day (1.3 mgd) (App. A, Fig. A6). Following the 16-day operation cycle, total volume of effluent applied during 1973 was almost 27.1 m.

Precipitation recorded at the treatment site during 1973 was about 112 cm (44 in.). Potential evaporation was estimated at about 66 cm/yr (26 in.), which was apportioned monthly by the ratio of monthly pat

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Table IX. Chemical and bacteriological characteristics of Imhoff tank effluent and annual wastewater additions to treatment beds (1973).

| Parameter | Effluent* | | Annual loadings† |
	Range	Mean	(kg/ha)**
pH (standard units)	6.2-8.0		
Conductivity (µmhos)	402-700	511	
Alkalinity (ppm $CaCO_3$)	116-245	155	42,560
Hardness (ppm $CaCO_3$)	22-60	41	12,260
BOD_5	30-185	112	30,750
COD	110-450	192	52,720
Total Nitrogen	19-78	47	12,910
Organic Nitrogen	11.5-32.8	23.4	6,425
NH_4-N	6.2-42	21.4	5,880
NO_3-N	0.4-2.8	1.3	360
NO_2-N	0.002-0.06	0.02	5.0
Total PO_4-P	6-16	11	3,020
Ortho PO_4-P	3-15	9	2,470
Chloride	75-210	150	41,190
Sulfate	27-72	42	11,530
Total Coliform bacteria $\times$ 10^6/100 ml	18-53	32	

* mg/l unless otherwise indicated.

† Based on 5,049 m^3/day effluent flow volume.

** kg/ha $\times$ 0.8929 = lb/acre.

evaporation to total annual pan evaporation (Table VIII).[26] While this method may not portray actual water loss from the treatment site, it provides a reasonable estimate. Summing monthly precipitation and potential evaporation showed that net monthly water loss from the site during June through September was 1.0 to 5.7 cm/mon, while gains during October through May were 1.6 to 9.6 cm/mon. Water gains and losses resulting from precipitation and evaporation were generally 1 to 2% of the total water additions to the treatment site; however, the above-normal precipitation during December accounted for about 11% of monthly total water.

Elevation of water tables in 14 observation wells varied during the hydrologic year. Water tables observed in wells 3, 8, and 12 illustrate the annual cycle observed in most wells (Fig. 7). Water tables were closest to the soil surface during early spring but decreased to a minimum elevation during late fall.

The regional groundwater flow was generally towards the Nashua River and adjacent surface water bodies. The groundwater gradient between well 2 and the river was generally 1.0 to 1.6%, while gradients from the other wells to the river or adjacent surface water were about 1%. Water tables appeared to vary directly with water level in the river, which was approximately, according to the water table of well 3, about 74 m from the river.

Well 1, located west of the treatment site, contained a measurable water table only from early spring to mid-summer. The water table elevation was 65 to 55 m (above mean sea level) between April and early July, before becoming dry.

The water table in well 2 remained relatively high throughout the study year, fluctuating from 64.5 to 65.6 m. While the water table elevation may have been influenced by the river flow stage, it apparently was more affected by which treatment beds were inundated at a particular time. When beds in close proximity of well 2 were used, the water table was high, but decreased in elevation when beds more removed were utilized.

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**Table X. Chemical and bacteriological characteristics of groundwater
in selected observation wells (1973).
Average values.**

Parameter*	Well										
	1	2	3	7	8	9	10	11	12	13	14
pH (standard units)	73	6.8	6.3	6.4	6.6	6.5	6.1	6.3	6.6	6.2	6.5
Conductivity (μmhos)	133	371	360	405	443	310	333	305	71	36	327
Alkalinity (as $CaCO_3$)	29	120	28	37	58	67	14	53	29	17	49
Hardness (as $CaCO_3$)	12	23	44	71	44	32	30	31	17	6	30
BOD_5	3.5	12	2.5	2.1	2.0	1.4	0.9	0.8	1.2	1.0	0.9
COD	42	26	19	22	15	8	10	9	10	13	11
Total Nitrogen	1.3	14.5	19.5	28.0	19.8	10.4	20.3	12.1	3.7	1.9	9.7
Organic Nitrogen	0.5	8.3	2.3	3.4	4.2	3.7	1.2	1.0	0.8	1.2	1.5
NH_4-N	0.6	5.3	1.3	4.7	4.5	3.2	0.5	1.0	0.3	0.3	0.4
NO_3-N	0.2	0.9	15.6	19.5	10.7	3.5	18.6	10.1	2.6	0.4	7.8
NO_2-N	0.01	0.03	0.3	0.4	0.4	0.02	0.02	0.02	0.01	0.01	0.02
Total PO_4-P	0.4	5.9	0.9	0.8	0.8	1.4	1.3	1.9	0.6	0.7	1.1
Ortho PO_4-P	0.1	5.6	0.2	0.1	0.1	0.4	0.1	0.2	0.1	0.1	0.1
Chloride	20	85	230	257	220	144	257	221	40	15	162
Sulfate	9	48	39	46	36	33	35	44	9	7	46
Total Coliform bacteria (no./100 ml)	335	3900	210	110	158	230	620	130	120	370	120

* mg/l unless otherwise indicated.

Chemical and bacteriological analyses of the waste-water effluent and the loadings of wastewater constituents to the treatment beds in 1973 are summarized in Table IX. The pH of primary sewage effluent was 6.2 to 8.0. The weekly additions of type-S lime to Imhoff tanks was probably responsible for the higher effluent pH values. The groundwater pH was 6.1 to 6.7 (Table X).

Wells 1 and 13 showed chloride levels were from 8 to 20 mg/l, which were representative of chloride levels of regional native groundwater. Motts and Saines[19] reported that chlorides in groundwater aquifer used for municipal and domestic water supplies, hereafter called native groundwater, generally range from 10 to 15 mg/l with some instances of the range 30 to 40 mg/l. Chloride levels in well 12, located 75 m from the treatment site, ranged from 12 to 25 mg/l throughout most of the hydrologic year. However, beginning in mid-November through December, chloride concentrations in well 12 increased to 120 mg/l. This increase was associated with a decline in water table elevation. Increased chloride levels reflect the influence of percolate from the treatment beds upon water quality and

the direction of groundwater flow from the site. Although the water table in well 13 declined during this period, chloride levels did not subsequently increase, as observed in well 12. Chlorides in wells 9 and 14 were of a similar level, with mean values of 144 and 162 mg/l, respectively. Chlorides in wells 3, 7, 8, 10, and 11 were the highest levels observed, with mean values of 220 to 280 mg/l.

Although annual chloride additions to the treatment beds were about 41,000 kg/ha yr, the chloride levels in the soil sample taken from the surface horizons of the treatment beds contained 1 to 8 ppm chloride more than background levels (Table V). Thus, since the treatment beds had been last excavated, chloride levels in the replacement soil had increased 2.5 to 4.4 kg/ha m. Since annual chloride inputs have far exceeded observed levels, the majority of chlorides have undoubtedly been carried through the soil to groundwater by percolating water. Because chlorides are readily carried through the sand and gravel strata, it is reasonable to assume that chloride concentrations in observation wells are indicative of the degree to which percolate from the treatment site affects

groundwater quality. In this regard, the water quality of wells 1 and 13 appeared unaffected, while that of the other wells was affected by the percolate. Well 12 appeared only slightly affected most of the year, but impacts increased during the latter portion of hydrologic year when groundwater gradients toward well 12 changed. The water quality in wells 2, 9, and 14 was moderately affected, while that in wells 3, 7, 8, 9, and 10 was more strongly affected.

It is uncertain at this time why chloride levels observed directly beneath the treatment site (well 2) averaged less than those in peripheral wells. One explanation is that native groundwater had mixed with the percolate.

Sulfate levels in groundwater also point out the relationship of the impact of the percolate on groundwater quality. Effluent sulfate concentrations ranged from 27 to 74 mg/l, with a mean of 42 mg/l. Annual sulfate inputs totaled about 12,000 kg/ha, which was not accounted for in the sulfate concentration of soil samples (Table IX). Sulfates in wells 1, 12, and 13 were 7 to 9 mg/l, while other observation wells had substantially larger levels.

The chemical oxygen demand (COD) of the primary effluent varied seasonally, 110 to 450 mg/l, with a mean value of 192 mg/l. Samples taken from wells 9, 10, 11, 12, and 14 showed COD levels slightly less than those observed for native groundwater, 8 to 11 mg/l. Wells most impacted by percolate from the treatment area had COD's higher than native groundwater (0.2 to 3.5 mg/l). Observed levels in wells 2, 3, 7, and 8 ranged from 15 to 26 mg/l, or about 8 to 14% of mean effluent levels.

Five-day biochemical oxygen demand (BOD_5) levels for the primary effluent were from 30 to 185 mg/l with a mean of 112 mg/l (Table IX). Although effluent BOD_5 was high, the groundwater surrounding the application site showed that BOD_5 had been effectively removed. Native groundwater BOD_5 levels varied from 0.2 to 3.5 mg/l, with a mean of 1.0 mg/l. Average levels in the groundwater directly beneath the application area (well 2) ranged from 8 to 18 mg/l with mean of 12 mg/l; however, these levels decreased as the percolate moved away from the treatment site. Groundwater BOD_5 in wells 9, 10, 11, 12, and 14, located 80 to 145 m from the treatment area, were generally less than 1.5 mg/l, while average BOD_5 levels in wells 3, 7, and 8 were 2.5 mg/l or less. Comparisons between the effluent and groundwater BOD_5 levels showed the

groundwater levels were generally less than 2% of the mean effluent concentration.

Calculation of total additions of organic constituents, expressed as BOD_5 and COD, showed that approximately 31,000 kg/ha yr of BOD_5 and 53,000 kg/ha yr of COD were added to the treatment beds (Table IX). These large additions suggest that organic matter may have accumulated in sufficient amounts to reduce water movement into and through the treatment beds. Analysis of soil samples from treatment beds showed that the annual accumulation increase of organic matter averaged about 0.2%/yr. This accumulation also included annual additions of the plant material grown naturally on the treatment beds. Although organic matter additions from plants growing on the beds has not been quantified, the relatively small percentage of increase of organic matter indicates that biological degradation of organic material has been quite effective. Levels of BOD_5 and COD in groundwater further bear this out. Groundwater BOD_5 and COD levels directly beneath and surrounding the treatment site were quite low and, in the wells most impacted by percolate COD and BOD_5, were slightly above native groundwater levels.

Total effluent nitrogen, which varied seasonally from 19 to 78 mg/l, appeared to be equally organic nitrogen and NH_4-N, with small amounts of NO_3-N and NO_2-N, 1.3 and 0.02 mg/l, respectively. Total nitrogen in the effluent was inversely related to wastewater flows: low flows, which occurred in late summer through early fall, had high nitrogen levels, while high flows, which occurred in late winter through spring, contained smaller nitrogen levels (Fig. 8). Seasonal variations were believed to reflect the dilution of wastewater by groundwater infiltration into sewerlines and the inflow of surface runoff through manholes which occasionally became inundated. Calculations show total nitrogen additions in 1973 were about 13,000 kg/ha.

Comparing nitrogen levels in effluent with those observed in groundwater showed groundwater nitrogen levels were 21 to 43% of effluent levels. Although the groundwater nitrogen levels were substantially less than the effluent concentrations observed, they were greater than average native groundwater level, 2.0 mg/l. Groundwater directly beneath the treatment area contained about 14.5 mg/l nitrogen, while wells which are peripheral to the treatment site contained 10 to 28 mg/l total nitrogen. Wells 9, 12, and 14, positioned
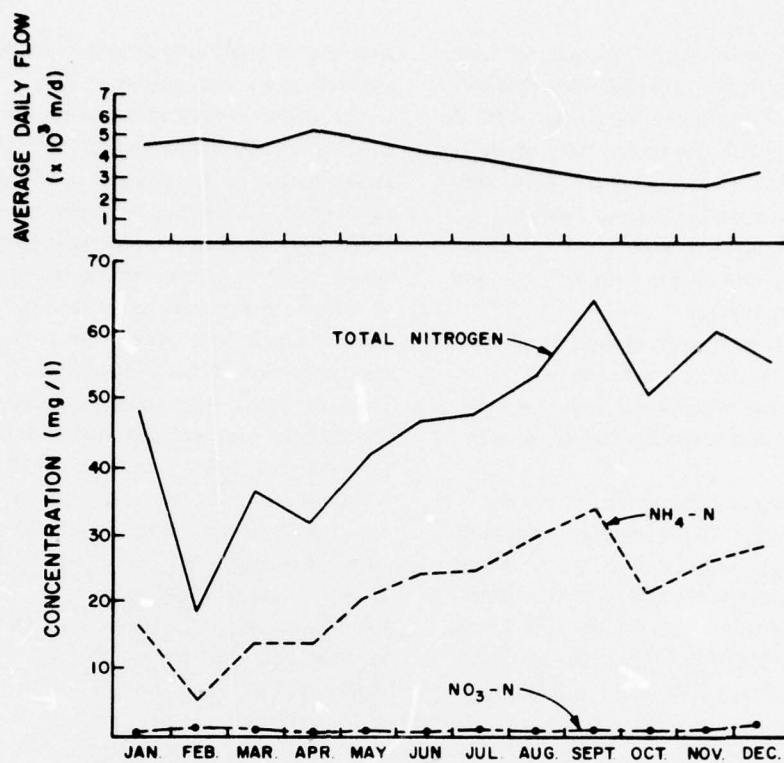
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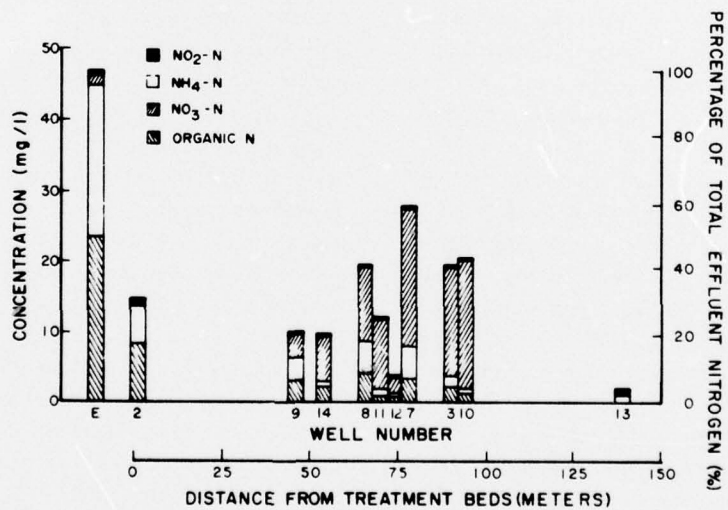Figure 8. Total nitrogen, ammonium nitrogen and nitrate nitrogen in primary sewage effluent (1973).

Figure 9. Concentrations of nitrite nitrogen, ammonium nitrogen, nitrate nitrogen and organic nitrogen in primary effluent and groundwater (1973).
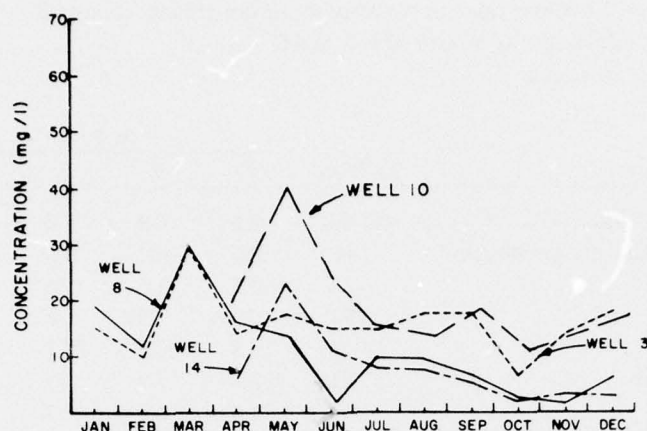
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Figure 10. Nitrate nitrogen in groundwater of selected wells (1973).

about 85-m distance from the treatment beds, had total nitrogen levels averaging about 10 mg/l. Wells 3, 7, 8, 10 and 11 contained the highest total nitrogen levels observed, 17 to 28 mg/l. Nitrogen levels detected in well 7 were believed to reflect the additional effects of leachate from adjacent sludge dewatering beds on groundwater quality.

Organic nitrogen ranged from 11 to 33 mg/l in the effluent, whereas concentrations in groundwater (wells 3, 7, 8, and 9) were only 2 to 4 mg/l greater than native groundwater levels, 1 to 2 mg/l. Effluent NH_4-N levels in wells 2, 7, and 8 were generally about 5 mg/l. Wells 3, 9, 10, 11, and 12 had less than 4 mg/l NH_4-N and native groundwater had about 0.3 mg/l NH_4-N.

NO_3-N composed the major portion of total nitrogen in groundwater of wells affected by percolate from the infiltration area and sludge beds (Fig. 9). NO_3-N in wells 3, 7, 8, 10, and 11 varied from 10 to 28 mg/l, although seasonal fluctuations were observed. Highest NO_3-N levels in wells 3, 8, 10, and 14 occurred during March through May, and low levels during October and November (Fig. 10). Peaks in NO_3-N curves during March and May reflected microbial degradation and nitrification processes of nitrogenous organics due to warmer temperature. NO_2-N was generally less than 0.5 mg/l in most wells, although concentrations of 0.5 to 1.0 mg/l were occasionally observed in wells 3, 7, and 8.

Changes in effluent nitrogen after application and percolation through 18 m of stratified sands and gravels, and resultant nitrogen levels in groundwater provided indirect evidence that nitrification had occurred. Effluent nitrogen was primarily organic nitrogen and NH_4-N with small amounts of NO_3-N and NO_2-N, but groundwater nitrogen was predominantly NO_3-N with small quantities of organic nitrogen and NH_4-N.

Several mechanisms for nitrogen removal were considered in view of groundwater nitrogen concentration compared with effluent concentrations. Mixing of the percolate with native groundwater, biological assimilation, soil adsorption, and denitrification could each have accounted for some nitrogen reduction. Dilution of percolate from the treatment site with native groundwater was possible; however, this was not substantiated by the chloride and sulfate levels and concentrations of nitrogen observed in wells 3, 4, 5, and 6. These wells were positioned at increasing 1.5-m (5-ft) increments to determine water quality changes through the 6.1-m (20-ft) depth. Chlorides in well 3 averaged 257 mg/l, and chlorides in wells 4, 5, and 6 were about 155 mg/l. Sulfate levels were similar for all wells (Table XI).

Comparison of chlorides and sulfate levels indicates that water quality was comparable to levels observed in effluent. This suggests little or no mixing of the percolate and native groundwater. Information showed saturated thickness beneath the beds (well 2) to be about 11.4 m (37.5 ft). Since saturated thickness in well 12 was about 4.2 m (14 ft), and "refusal" sloped toward the river beneath the treatment beds, native groundwater was probably moving beneath the treatment site at an elevation below that found

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Table XI. Chemical characteristics of primary effluent compared
with water quality in wells 3, 4, 5, and 6.
Average values.

Parameter*	Effluent	Well 3	Well 4	Well 5	Well 6
pH (Standard Units)	6.2-8.0	6.3	6.9	7.0	7.1
Electrical conductivity (μmhos/cm)	511	360	441	502	466
BOD_5	122	2.5	1.5	1.6	1.9
COD	192	19	29	28	34
Total Nitrogen	47	19.5	20.6	22.1	22.2
Organic-nitrogen	23	2.8	7.9	11	8.0
NH_4-N	21	1.3	6.4	7	8.4
NO_3-N	1.3	15.6	5.9	3.8	5.7
NO_2-N	0.02	0.3	0.4	0.3	0.1
Total PO_4-P	11	0.9	0.9	0.8	1.0
Ortho PO_4-P	9	0.2	0.3	0.1	0.2
Cl	150	257	157	159	155
SO_4	42	46	43	47	46

* mg/l unless otherwise noted.

beneath wells 3, 4, 5, and 6. Chlorides and other parameters determined for wells 3, 4, 5, and 6 were comparable; thus, mixing did not appear to be an important consideration for observed differences in nitrogen. Mixing, which may have occurred, would probably have been very small because distances between the wells were short; in addition the soil particles would have constrained mixing in sand and gravel strata.

Nitrogen reduction from dilution by precipitation could create low nitrogen concentrations for short periods, but gains from precipitation inputs were calculated to be only 2% of the monthly net water budget. This small percentage could not account for differences between the nitrogen levels in the effluent and those in the observation wells. Dilution would most probably be of short duration, but this was not apparent from NO_3-N data plotted in Figure 10.

Biological assimilation of the nitrogen by bacteria and other microorganisms accounted for some nitrogen removal. The uptake of nitrogen by grasses and other surface macrophytes growing on the filter beds removed about 0.4% of the nitrogen applied annually. However, because surface plants and microbes were not removed periodically, assimilated nitrogen was released back into the percolate at a later time through degradation of nitrogenous organic compounds. Slowly, but

eventually, all nitrogen would be released back to the system.

Fixation of nitrogen in relatively stable organic matter with subsequent adsorption of NH_4-N could account for substantial nitrogen removal.[12] Nitrogen levels in milligrams per gram of soil in the surface samples of filter beds 13, 14, and 15, after receiving primary effluent for five years, were 0.06 to 0.4 mg/l more than background levels. Assuming that primary effluent quality since October 1968 was comparable to that observed during 1973, annual nitrogen accumulation was 83 to 134 kg/ha. This represents only 0.6 to 1.0% of the total annual nitrogen additions.

Adsorption of the NH_4-N on soil exchange sites could remove the NH_4-N applied in the effluent over a three-day period. Following the methodology outlined by Lance,[12] ammonium retained in the 1-m aerobic surface horizon was calculated. The concentrations of calcium and magnesium, the major competing divalent cation in the primary effluent, were approximately 205. and 4.84 ppm, respectively.

Ammonia adsorption ratio (AAR) can be calculated using the equation:

$$AAR = NH_4 / (½ Ca + Mg)^{½} \qquad (1)$$

where concentrations are expressed in milliequivalents

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per liter. Experimental data are not available for NH_4-N adsorption but have been obtained for K^+ adsorption in western soils.[24] Since K^+ and NH_4-N behave similarly in cation exchange reactions, the following equation, relating the potassium adsorption ratio (PAR) of a solution to the exchangeable potassium percentage (EPP) of the soil, can be used to estimate exchangeable ammonium percentage (EAP):

$$EPP = \frac{100 (0.0360 + 0.1051\ PAR)}{1 + (0.0360 + 0.1051\ PAR)} . \qquad (2)$$

From AAR (1.41), the exchangeable ammonium percentage was estimated using eq 2. For conditions present here, 15.6% of total exchangeable sites in the soil could be occupied by NH_4-N after effluent of the above composition equilibrates with the soil. Soil which had a mean cation exchange capacity of 1.22 meq/100 g of soil could contain about 0.034 mg of NH_4-N/g of soil according to these calculations. Thus, 1 ha-m of soil could adsorb NH_4-N in 1.47 m^3 (388 gal.) of primary effluent containing 21.4 mg/l of NH_4-N and the same concentrations of calcium and magnesium used above. This would be the amount of NH_4+ applied to three 0.32-ha beds over a three-day period where effluent flow was 5061 m^3.

To rejuvenate the ammonium adsorptive capacity, adsorbed ammonia must be removed from the treatment bed sometime prior to the next application period. Lance and Whisler[13] observed in laboratory column studies that ammonium retained on the soil exchange complex in filter columns was nitrified during the dry period where aerobic conditions existed. If NH_4+ adsorption capacity was not rejuvenated, additional ammonium applied during the next application period would be carried through the treatment beds into the groundwater.[7]

Ammonium retained on the exchange capacity would nitrify during the ensuing recovery period and create a high concentration of nitrate in water held in the soil. During the initial segment of the next inundation period, the NO_3-N would be leached through the sand and gravel material, creating nitrate peaks in the groundwater. No such peaks were observed during 1973; this probably resulted from the rotation cycle of filter bed inundation which would attentuate any nitrate peaks. Increasing the sampling frequency would possibly result in the detection of any existing nitrate peaks.

It is uncertain whether nitrogen levels observed in the groundwater resulted from denitrification. However, in view of the large annual nitrogen inputs over the past 30 years and the limited capacity of other nitrogen removal mechanisms observed, nitrogen reduction could be accomplished only through the denitrification process.

The presence of various nitrogen forms in groundwater is proof that nitrification of organic and ammonium nitrogen has occurred. The concentrations of organic and ammonium nitrogen in groundwater samples were about 17 and 20%, respectively, of the effluent levels, whereas NO_3-N levels increased 8 to 15-fold in groundwater in wells.

Not only must nitrogen be in the proper form (NO_3-N) for denitrification to occur, but an ample available source of organic carbon must be present for both nitrification and denitrification to occur. St. Amant and McCarty[25] determined that 1.3 mg of carbon supplied as methanol is required for every milligram of nitrogen reduced to nitrogen gas. Using the equation

$$\frac{BOD}{TOC} = \frac{32}{12} (0.9)(0.77) = 1.85 \qquad (3)$$

the amount of total organic carbon (TOC) in the primary sewage effluent was approximated where the ultimate BOD equaled 90% of the theoretical oxygen demand and 5-day BOD was 77% of the ultimate BOD for domestic wastes.[9]

The C:N ratio was calculated to the range 0.85 to 1.29. The actual ratio may have been higher than that calculated, since about half of the nitrogen was in organic form and grasses, algae and other plants growing on the treatment beds added organic material which is not removed annually and which would add to the supply of available carbon.

These calculations show that sufficient organic carbon was present at the surface of the treatment beds to facilitate denitrification provided other environmental conditions were amenable. The high nitrate in the groundwater shows that optimum conditions for denitrification did not exist. Findings at the Flushing at the Flushing Meadows Project[7] suggest the 2-day treatment-bed inundation practiced at Fort Devens was adequate for nitrification, but was not long enough to enhance denitrification. Lance and Whisler[13] found that a 2-day inundation, followed by a 5-day recovery period, was ideal for nitrifying organic

and ammoniacal nitrogen but did not remove nitrogen from the groundwater. Increasing the inundation period to 9 days, followed by 5 days recovery, resulted in substantial nitrogen reduction which Lance and Whisler attributed to denitrification.[13] Since sufficient organic carbon appeared to be available and the nitrification process was operating, adjustments in the inundation period could have resulted in lower nitrogen levels in the groundwater.

Phosphates in the primary sewage effluent varied from 6 to 16 mg/l with an average concentration of 11 mg/l. Calculating phosphate additions to the treatment site showed that, although substantial amounts of phosphates were applied, groundwater levels remained only slightly higher than background levels. Phosphates in the native groundwater (well 13) averaged about 0.7 mg/l total PO_4-P, 0.1 mg/l of which was ortho-PO_4-P or available PO_4-P. Groundwater in the observation wells generally contained less than 2 mg/l total PO_4-P and 0.1 to 0.4 mg/l ortho-PO_4-P. Phosphate levels beneath the treatment site were 6 mg/l total PO_4-P, 5.6 mg/l of which was ortho-PO_4-P. However, these levels decreased in the surrounding area to less than 2 mg/l total PO_4-P at a distance of 80 to 150 m.

Soil data obtained from treatment beds showed that considerable phosphorus was contained in various strata. While soil phosphorus levels varied with depth and between treatment beds, total phosphorus was 5,570 to 7,060 kg/ha-m. Thus, phosphorus adsorptive capacity was quite large; this would account for the low phosphorus levels in the groundwater, which were about 11% of effluent levels.

Infrequent analysis for fecal coliform bacteria proved negative for all groundwater samples. Total coliform bacteria in the effluent varied from 18×10^6 to 53×10^6/100 ml of effluent with a mean coliform level of approximately 32×10^6/100 ml (Table IX). Bacterial analysis of the groundwater samples showed that mean total coliform bacteria in well 2 was about 4,000/100 ml but was generally less than 200/100 ml in the other wells. Because none of the well casings had been sealed, there was a good possibility that the major portion of the total coliform bacteria observed in the wells was indigenous coliform bacteria and did not originate from the effluent.

Under the 2-day inundation, 14-day recovery cycle, impacts of percolate resulting from the rapid infiltration treatment basins at Fort Devens did not substantially impair groundwater quality in the surrounding area. While concentrations of most water quality

parameters had increased to varying degrees, only NO_3-N and NH_4-N levels were sufficient to cause real concern. BOD, COD, electrical conductivity, phosphorus, sulfate, chlorides and total coliform bacteria were found at somewhat higher levels than observed in native groundwater quality, but were not high enough to warrant major changes in pretreatment of wastewater.

Groundwater nitrogen levels, particularly NO_3-N and NH_4-N, warrant some changes in the operational procedures. While more sophisticated methods could be implemented to reduce groundwater nitrogen levels, nonstructural procedures should be evaluated, particularly adjustments in the application cycle. Operational changes are those that would enhance nitrification and denitrification processes so that organic nitrogen and NH_4-N would be denitrified to nitrogen gas and volatilized to the atmosphere.

Factors of the nitrification and denitrification processes have been enumerated.[1] Because most effluent nitrogen is in the ammoniacal and organic form, both nitrification and denitrification would have to occur simultaneously in the same treatment bed. Basic differences between the two processes are the form of nitrogen being converted, the oxygen conditions in the microenvironment where the process occurs, and the organism necessary to carry out the transformations. Oxidation of organic and ammoniacal nitrogen to nitrate requires aerobic conditions, whereas dissimilation of nitrate to nitrogen gas requires anaerobic conditions. Presently, the application rate and short inundation period, followed by a long recovery period, are well suited to nitrification. Anaerobic conditions probably do not exist in the treatment beds long enough to optimize nitrate reduction. To reduce NO_3-N levels in groundwater, the length of the inundation period should be increased to enhance the denitrification of nitrate nitrogen. Carbon scarcity should not be a problem because of the additions of organic carbon to the primary effluent and the plant detritus to the treatment beds.

Changes in the application cycle could result in additional nitrogen removal through nitrification and denitrification, but extensive field investigation will be necessary to define the management conditions required at the Fort Devens facility.

The percolate from the beds has resulted in an increase in the electrical conductivity, nitrogen, chloride and sulfate levels. Although these constituents were increased in the native groundwater, the quality

of the groundwater peripheral to the treatment sites continues to meet Public Health Service[22] drinking water standards, with the exception of NH_3-N and NH_4-N levels.

Even so, the quality of percolate from the rapid infiltration facility at Fort Devens was better than the quality of effluent reported for conventional tertiary wastewater treatment facilities, consisting of extended aeration activated sludge with alum polyelectrolyte coagulation, sedimentation, and filtration.[16]

CONCLUSIONS

1. The rapid infiltration system has operated effectively throughout the year, even though greater ponding of effluent was observed during winter operation than during warmer periods of the year.

2. The 2 days wet/14 days drying operation cycle has maintained the infiltration and percolation capacities of the treatment beds after six years of field operation.

3. The levels of nitrogen, phosphorus, calcium, magnesium, potassium, iron, sodium, copper, zinc, chloride, and sulfur have increased in the soil surface horizon of the treatment beds.

4. The percolate from the treatment site has increased the levels of nitrogen, chloride, sulfate, conductivity, alkalinity, and hardness in the groundwater. Slight increases in BOD, COD, and total coliform bacteria levels have been observed.

5. Differences in the concentration of organic-nitrogen, NH_4-N, NO_3-N, and NO_2-N between the primary effluent and the groundwater illustrate that ammonification and nitrification were occurring in the treatment beds. A total nitrogen balance showed a 60 to 80% reduction in total nitrogen applied to the treatment beds.

6. Because of the limited capacities of the other plausible nitrogen removal mechanisms and the substantial quantities of nitrogen removed, the reduction in total nitrogen levels observed may be attributable to denitrification.

7. The groundwater quality in the strongly impacted wells indicates that the unchlorinated primary sewage effluent is being renovated to a quality comparable to that achieved by conventional tertiary wastewater treatment facilities.

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APPENDIX A. ADDITIONAL SOIL, WASTE-WATER FLOW AND PRECIPITATION DATA FOR THE FORT DEVENS WASTEWATER TREATMENT FACILITY

INTRODUCTION

Daily wastewater flows to the treatment facility were obtained from operators' logs kept at the treatment facility and by the Facilities Engineer, Fort Devens, Massachusetts. Seven-day average flows were the sum of daily flows recorded during a 7-day period divided by 7. Maximum and minimum flows for each 7-day period were the recorded largest and smallest daily flows during the 7-day period.

Climatic data were collected at the facility in the past by measuring temperatures once in the morning and again in the afternoon. Precipitation events and wind direction were noted, but measurements were not taken. Precipitation data reported in Figures A1-A6 were obtained from Weather Bureau data for Fitchburg, Massachusetts, located 12.9 km (8 miles) west-southwest of the Fort Devens treatment facility. During 1973, precipitation occurring at the treatment site was recorded using a U.S. Weather Bureau rain gage. Compared with daily precipitation measurements reported for Fitchburg, differences were noted between specific events but monthly and yearly totals were approximate.

Figure A1. Maximum, minimum and average 7-day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1968).

Figure A2. Maximum, minimum and average 7-day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1969).

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Figure A3. Maximum, minimum and average 7-day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1970).

Figure A4. Maximum, minimum and average 7-day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1971).
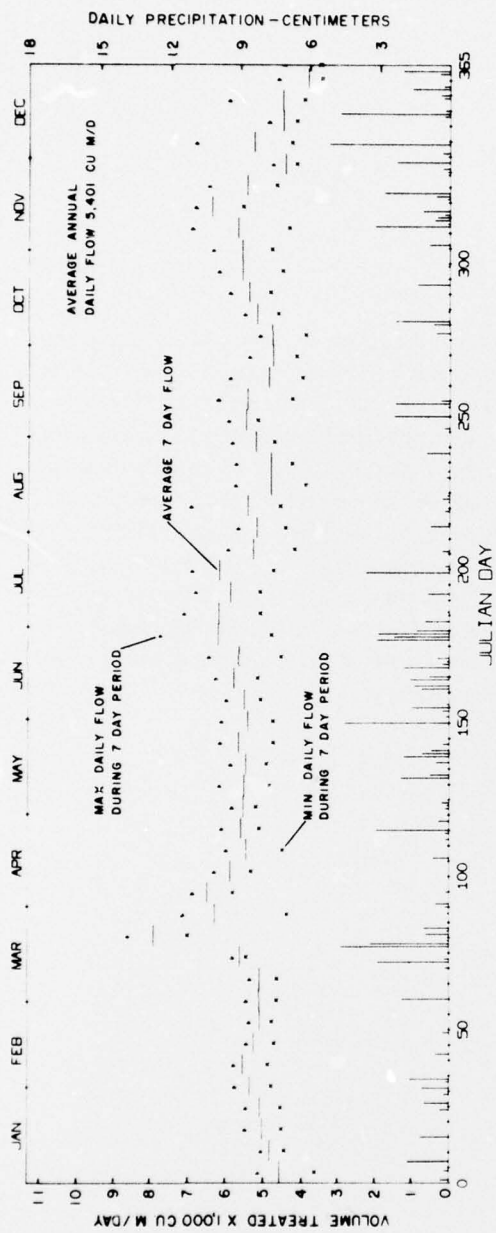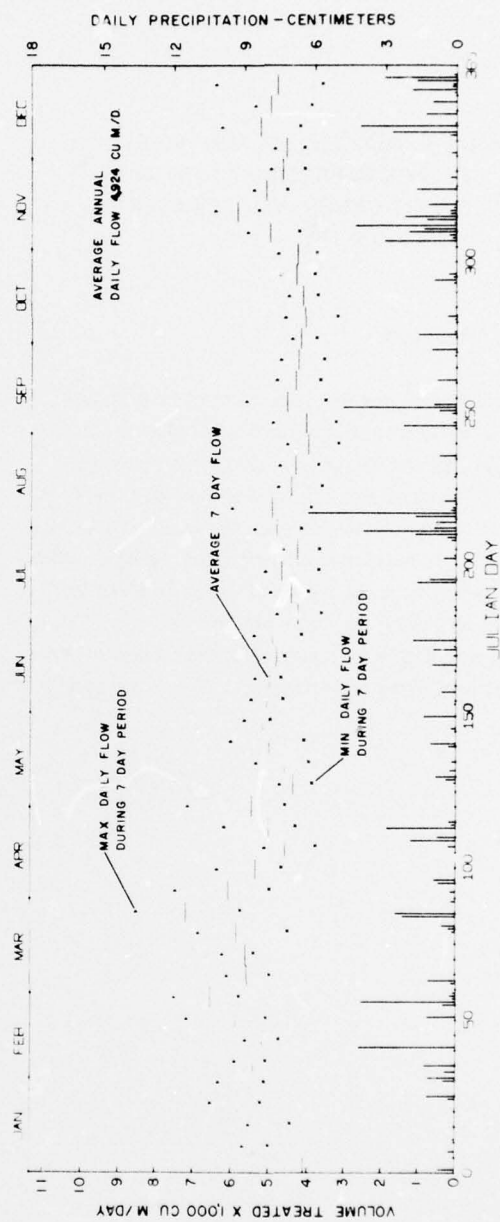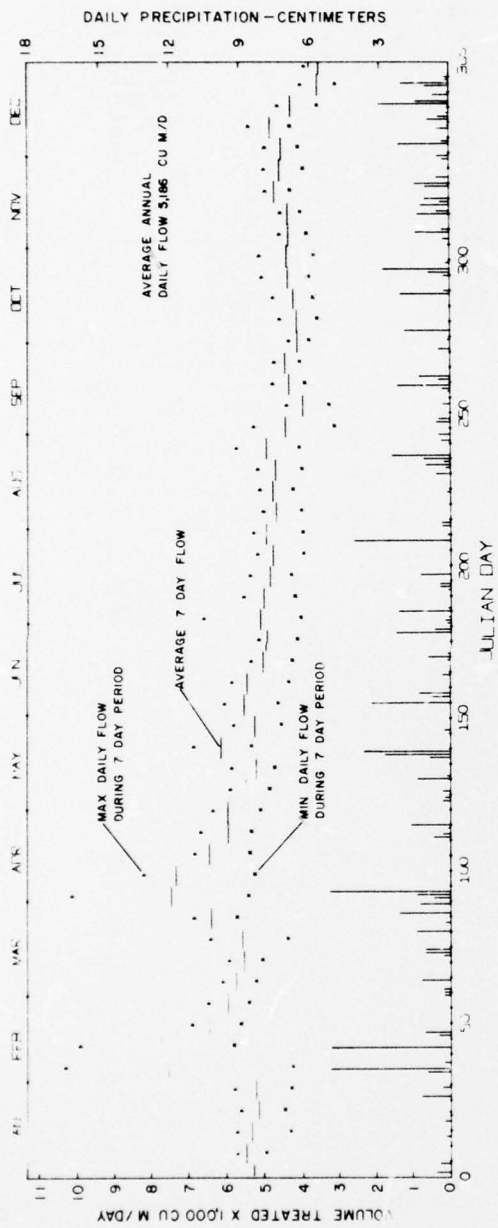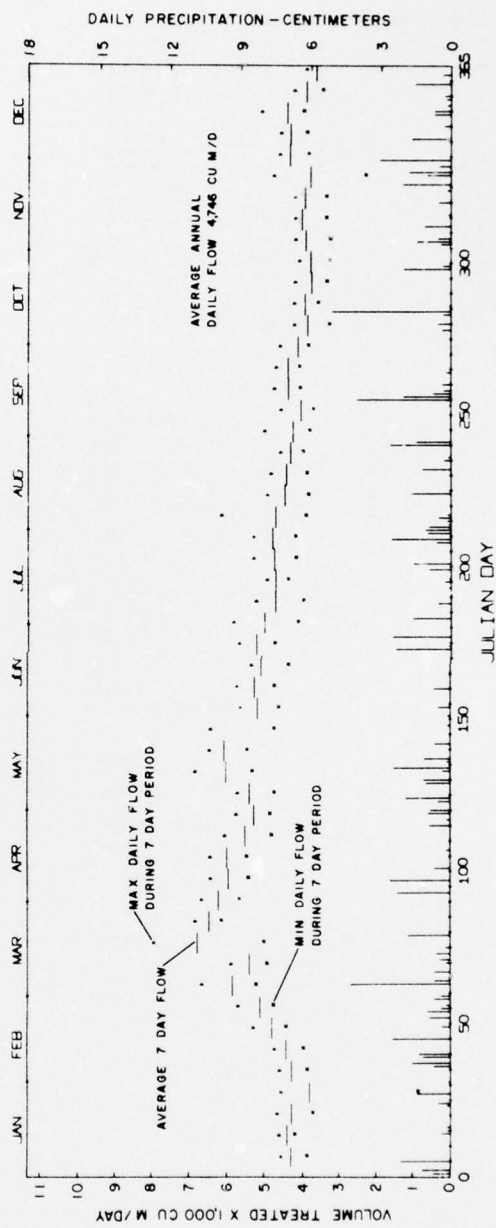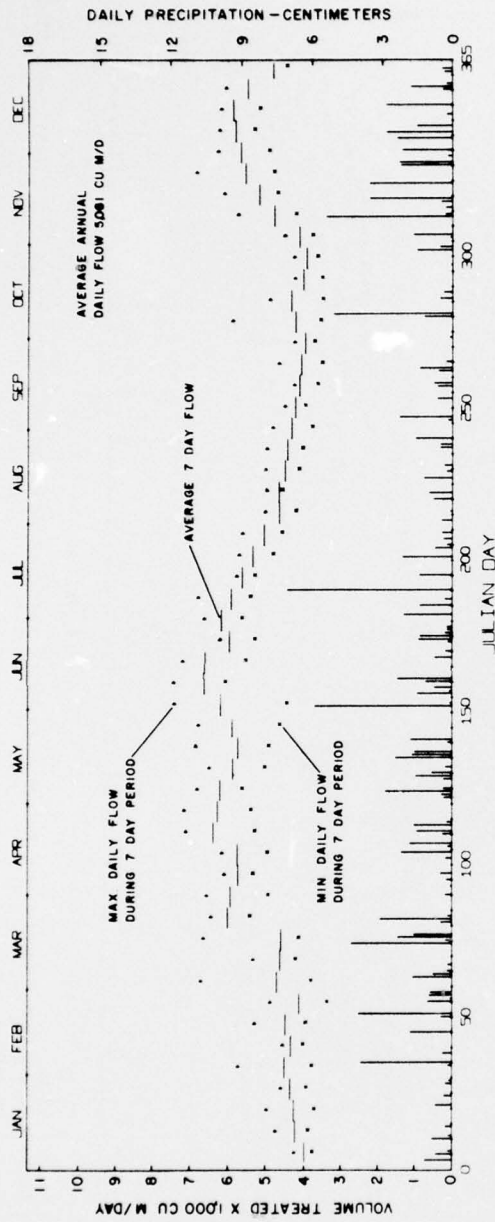
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Figure A5. Maximum, minimum and average 7-day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1972).
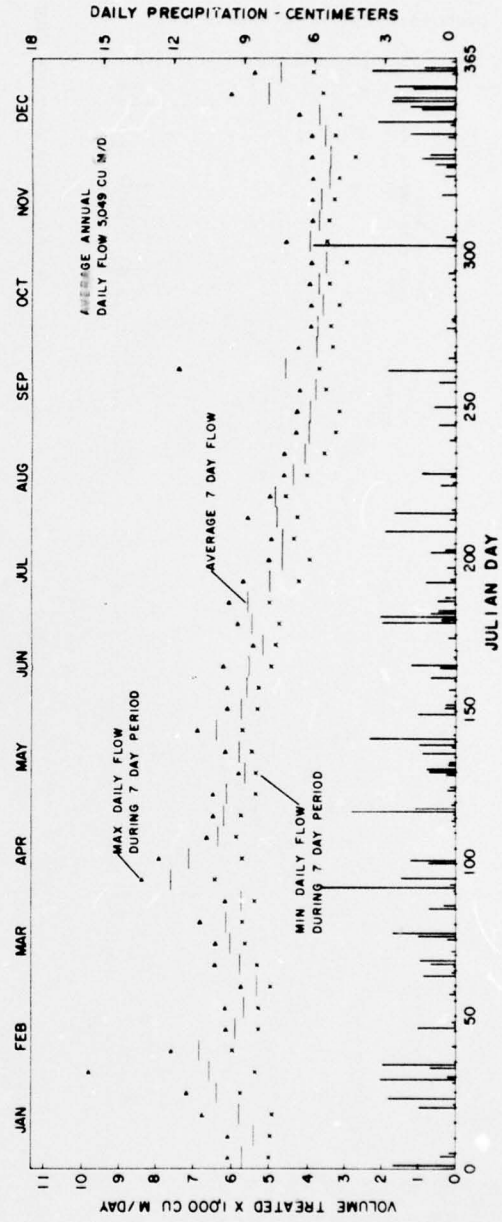

Figure A6. Maximum, minimum and average 7-day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1973).

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Table AI. Sieve analysis of treatment bed soil samples.

Sample depth (m)	Percentage of sample finer than sieve openings of					
	2.00 (mm)	1.00 (mm)	0.50 (mm)	0.25 (mm)	0.10 (mm)	0.053 (mm)
Filter bed 13*						
0-0.15	100.0	86.0	42.0	10.0	1.0	1.0
0.15-0.46	100.0	88.0	49.0	17.0	4.0	2.0
0.46-0.61	100.0	82.0	40.0	14.0	3.0	1.0
0.61-0.91	100.0	80.0	40.0	16.0	5.0	2.0
0.91-1.52	100.0	83.0	48.0	26.0	12.0	6.0
1.52-1.98	100.0	88.0	49.0	21.0	6.0	3.0
1.98-2.59	100.0	88.0	55.0	37.0	11.0	6.0
2.59-3.05	100.0	89.0	63.0	41.0	22.0	11.0
Filter bed 14						
0-0.30	56.0	46.0	28.0	12.0	4.0	1.0
0.30-0.61	58.0	48.0	31.0	10.0	3.0	2.0
0.61-1.22	62.0	46.0	23.0	8.0	4.0	2.0
1.22-1.52	34.0	28.0	19.0	10.0	5.0	3.0
1.52-2.74	63.0	52.0	28.0	12.0	6.0	3.0
2.74-3.05	38.0	30.0	20.0	13.0	7.0	4.0
3.05-4.57	55.0	37.0	20.0	12.0	6.0	3.0
4.57-4.88	98.0	96.0	88.0	25.0	6.0	3.0
4.88-4.97	41.0	34.0	25.0	14.0	7.0	4.0
4.97-5.64	94.0	81.0	44.0	22.0	11.0	6.0
5.64-6.10	77.0	64.0	35.0	16.0	8.0	4.0
Filter bed 15						
0-0.15	74.0	62.0	38.0	18.0	7.0	3.0
0.15-0.46	40.0	32.0	18.0	7.0	3.0	1.0
0.46-1.22	67.0	50.0	28.0	8.0	2.0	1.0
1.22-1.52	54.0	44.0	28.0	10.0	4.0	2.0
1.52-3.05	73.0	57.0	28.0	12.0	5.0	1.0
3.05-4.27	70.0	54.0	26.0	11.0	5.0	1.0
4.27-4.57	88.0	72.0	43.0	22.0	11.0	6.0
4.57-4.88	72.0	50.0	25.0	12.0	6.0	3.0
4.88-5.79	61.0	48.0	28.0	16.0	8.0	5.0
5.79-6.10	62.0	48.0	29.0	18.0	10.0	6.0
Filter bed 16						
0-0.46	78.0	64.0	36.0	14.0	4.0	1.0
0.46-0.76	52.0	43.0	23.0	9.0	3.0	1.0
0.76-0.91	63.0	53.0	29.0	8.0	3.0	1.0
0.91-1.52	41.0	32.0	19.0	9.0	4.0	2.0
1.52-1.68	88.0	71.0	41.0	19.0	9.0	4.0
1.68-2.13	78.0	56.0	26.0	11.0	5.0	2.0
2.13-3.05	51.0	41.0	22.0	10.0	6.0	3.0

* Particles greater than 2 mm not included in percentage determinations.

Table AI (cont'd). Sieve analysis of treatment bed soil samples.

Sample depth (m)	Percentage of sample finer than sieve openings of					
	2.00 (mm)	1.00 (mm)	0.50 (mm)	0.25 (mm)	0.10 (mm)	0.053 (mm)
Filter bed 16 (cont'd)						
3.05- 3.96	48.0	34.0	18.0	9.0	4.0	2.0
3.96- 4.57	78.0	51.0	21.0	10.0	4.0	2.0
4.57- 6.10	63.0	52.0	33.0	21.0	11.0	6.0
Well no. 2						
0- 0.24	82.0	71.0	46.0	18.0	8.0	5.0
0.24- 1.28	85.0	76.0	46.0	17.0	8.0	5.0
1.28- 1.52	47.0	40.0	24.0	12.0	5.0	3.0
1.52- 2.71	74.0	57.0	31.0	11.0	3.0	2.0
2.71- 3.05	59.0	46.0	28.0	14.0	7.0	4.0
3.05- 4.57	76.0	57.0	29.0	13.0	6.0	4.0
6.10- 7.62	68.0	49.0	27.0	15.0	8.0	4.0
7.62- 9.14	73.0	59.0	36.0	17.0	8.0	4.0
9.14-10.67	90.0	87.0	32.0	8.0	3.0	2.0
10.67-12.19	85.0	76.0	37.0	11.0	5.0	3.0
12.19-13.72	80.0	73.0	23.0	6.0	2.0	1.0
13.72-15.24	84.0	72.0	37.0	12.0	6.0	3.0
15.24-16.74	93.0	84.0	45.0	10.0	3.0	2.0
16.74-18.29	100.0	98.0	83.0	34.0	14.0	5.0
18.29-19.81	100.0	100.0	95.0	54.0	16.0	5.0
19.81-21.34	100.0	100.0	98.0	14.0	1.0	0.0
22.86-24.38	100.0	98.0	74.0	40.0	26.0	5.0
Background material						
0- 0.15	91.0	84.0	57.0	28.0	14.0	10.0
0.15- 0.30	85.0	78.0	58.0	38.0	22.0	15.0
0.30- 0.46	48.0	39.0	30.0	21.0	14.0	12.0
0.46- 0.61	71.0	60.0	31.0	6.0	1.0	0.0
0.61- 0.91	54.0	45.0	22.0	2.0	1.0	0.0
0.91- 1.83	95.0	87.0	62.0	18.0	1.0	0.0
1.83- 2.44	88.0	80.0	59.0	27.0	3.0	1.0
2.44- 3.05	99.0	99.0	96.0	60.0	4.0	0.0

* Particles greater than 2 mm not included in percentage determinations.